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The Royal Society of Canada Expert Panel:

The Behaviour and Environmental Impacts of Crude Oil Released into Aqueous Environments

Fall 2015

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Behaviour and Environmental Impacts of Crude Oil Released into Aqueous Environments

An Expert Panel Report prepared at the request of
the Royal Society of Canada
for the Canadian Energy Pipeline Association
and the Canadian Association of Petroleum Producers

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The Royal Society of Canada
La Société royale du Canada

November 2015

ISBN: 978-1-928140-02-3

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RSC: EPR 15-1

The report should be cited as follows:

Lee, Kenneth *(chair),* Michel Boufadel, Bing Chen, Julia Foght, Peter Hodson, Stella Swanson, Albert Venosa. (2015). Expert Panel Report on the Behaviour and Environmental Impacts of Crude Oil Released into Aqueous Environments. Royal Society of Canada, Ottawa, ON. ISBN: 978-1-928140-02-3

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of the report a few days before it is released so they have an opportunity to prepare responses to its content by the time the report is publically released.

6. **Ownership of the report.** The report belongs to the RSC . However, the RSC typically gives written permission to the panelists / authors to rework the document and submit portions of it to academic journals or other publications if they wish to do so.

Organizations or individuals interested in discussing opportunities for the RSC to carry out an Expert Panel on a topic of interest to them should contact the Chair of the Committee on Expert Panels at epsec@rsc-src.ca.

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Preface

Canadians are among the highest per capita consumers of oil in the world, equivalent to about 25 barrels—or about 4000 litres—of oil per person per year.

Canada is also one of the world's only developed economies that produces more oil than it demands for domestic consumption, currently about 1.5 times more (about 6000 L/person/yr). In recent years, the rate of oil production has been increasing by about 5% per year, primarily through the growth of Alberta's oil sands.

To meet both domestic and international demand, the oil must be moved by pipeline, rail, tanker trucks or ship from where it is recovered to where it will be refined and ultimately used. In recent years, the energy industries have been developing plans and seeking permission to expand or modify their transportation networks, especially those involving pipelines. Keystone XL, Northern Gateway, Trans-Mountain, Line 9 Reversal and Canada East are names of pipeline projects that have regularly been a regular feature in the news.

Hot-button topics at the forefront of those concerned with these pipeline projects include questions about the consequences of accidental spills of crude oil into aquatic ecosystems. *Do we have sufficient knowledge about how crude oils behave when released into fresh waters, estuaries or oceans to develop effective strategies for spill preparedness, spill response and remediation? What are the gaps in knowledge and how should research insights inform policies, regulations and practices in this area?*

Among the conditions for the approval of the Gateway project, the National Energy Board required the applicants to establish '*a research program regarding the behavior and cleanup (including recovery) of heavy oils spilled in freshwater and marine aquatic environments*' (Condition 169).[3] In mid 2014, the Royal Society of Canada's (RSCs) Committee on Expert Panels was approached about carrying out an independent, arm's length assessment of the state of the science in this area.

Since the prospective funders of this panel (Canadian Energy Pipeline Association (CEPA) and the Canadian Association of Petroleum Producers (CAPP)) might be perceived to have a strong interest in the outcome of the work, the Committee on Expert Panels was especially rigorous in following its procedures (see previous pages) to ensure an independent, balanced and objective report.

CEPA and CAPP suggested the initial Terms of Reference, and were receptive to changes in those terms following RSC preliminary consultation with diverse experts. Neither CEPA nor CAPP (or their membership) had any input into the selection of panelists or reviewers, or into the recommendations of this panel.

The RSC thanks CEPA and CAPP for their support of this project and appreciates their professional approach that let us go about our work without interference of any kind.

We wish to thank the Panel Chair, Dr. Ken Lee and his fellow panelists (listed on the next page) for volunteering their time and expertise to prepare this report. This comprehensive and high quality report did not emerge without exceptional effort. Thank you!

We also want to thank the Peer Review Monitor, Dr. Robie Macdonald, FRSC and eight expert reviewers (listed on the next page), who provided extensive comments, criticisms and suggestions on the first draft of this report. The additions and clarifications that resulted from their feedback significantly added to the quality of the document.

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[3] *Report of the Joint Review Panel for the Enbridge Northern Gateway Project (Dec 19, 2013) Conditions 169, 170, and 193* http://gatewaypanel.review-examen.gc.ca/clf-nsi/hm-eng.html

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Finally, a special thank you to Russel MacDonald from the RSC office in Ottawa and Lisa Isaacman from Bonnyville who provided administrative and editorial support for the work of the panel and to the members of the Committee on Expert Panels and its Oversight Committee that were involved in selecting the panelists and reviewers.

David B Layzell, PhD, FRSC
Chair, Committee on Expert Panels

Graham Bell, PhD, FRSC
President, Royal Society of Canada

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Panel Members

Dr. Kenneth Lee (Chair) is the Director of Oceans and Atmosphere , Commonwealth Scientific and Industrial Research Organisation (CSIRO) in Perth, Australia. This research group provides large-scale multidisciplinary science to support the management and sustainable development of Australia's marine and atmospheric resources. The founding Executive Director of the Centre for Offshore Oil, Gas and Energy Research (COOGER), Department of Fisheries and Oceans , Canada, he received the "Prix d'Excellence" for exemplary contributions in science (Fisheries and Oceans Canada) and the Government of Canada's "Federal Partners in Technology Transfer - Leadership Award" for technology transfer from a federal laboratory to the private sector.

Dr. Michel Boufadel is Professor of Environmental Engineering and Director of the Center for N atural Resources Development and Protection at the New Jersey Institute of Technology. He is a Professional Engineer in Pennsylvania and New Jersey, a Board Certified Professional Engineer, and a Fellow in the American Society of Civil Engineers.

Dr. Bing Chen is the leader of the Northern Region Persistent Organic Pollution Control (NRPOP) Laboratory, Associate Professor of Civil Engineering with the Faculty of Engineering and Applied Science, and Chair of the Environmental Systems Engineering & Management Program at the Memorial University of Newfoundland, St. John's, NL, Canada.

Dr. Julia Foght is Professor Emerita at the University of Alberta, where she was a Professor of Petroleum Microbiology in the Department of Biological Sciences from 1994 to 2014. Julia received the Petro -Canada Young Innovators Award in 2001, a McCalla Professorship in 2 011 from the University of Alberta, and in 2014 the Alberta Science & Technology Foundation (ASTech) award in Innovation in Oil Sands Research.

Dr. Peter V. Hodson is an ecotoxicologist and Professor Emeritus with Biology and the School of Environmental Studies at Queen's University, and a Past President of the Society of Environmental Toxicology and Chemistry. He began his career with Fisheries and Oceans and his research contributed to the development of Great Lakes Water Quality Objectives under the 1972 Canada-US Great Lakes Water Quality Agreement and to the development of Federal Environmental Effects Monitoring Programs for the pulp and paper sector. Since 1995, he has focused on the toxicity of petroleum hydrocarbons to fish embryonic development and the role of con taminants in the decline of American eel abundance in Lake Ontario.

Dr. Stella Swanson is an aquatic ecologist and risk assessment specialist with Swanson Environmental Strategies that she operates out of Calgary, AB and Fernie, BC. Her 30-year career has included management of the Aquatic Biology Group at the Saskatchewan Research Council, and consulting positions with SENTAR Consultants (now Stantec) and Golder Associates Ltd. (where she attained the position of Principal).

Dr. Albert D. Venosa is the former Director of the Land Remediation and Pollution Control Division (LRPCD), one of the five divis ions within the National Risk Management Research Laboratory (NRMRL), Cincinnati, OH, in the United States Environmental Protection Agency's (U.S. EPA's) Office of Research and Development. As an environmental scientist and microbiologist, he managed EPA's oil spill research program from 1990 until 2014, during which time he became an expert in bioremediation and dispersant research and other response technologies. Prior to that, he led U.S. EPA's national program on wastewater disinfection research for 13 years. He is now retired after 45 years of government service.

Support of the Work of the Panel

We are grateful to the following individuals:

Christopher Stone, (Calgary, Alberta) for his assistance in compiling references and identifying prospective panellists and reviewers.

Lisa Isaacman, (Bonnyville, Alberta) for her administrative work with the panel and editorial work on the report

Russel MacDonald , (Officer, Expert Panels & the College, the Royal Society of Canada , Ottawa, Ontario) for his assistance in managing the funds, helping with logistics, taking notes and pr oducing the final report.

Peter Christie (Science Writer, Kingston, O ntario) for his assistance with some critical sections of the report.

Peer Reviewers

The Royal Society of Canada would like to acknowledge and sincerely thank the Peer Review Monitor and eight peer reviewers who provided valuable, detailed input into the first draft of this report. The following reviewers have permitted the RSC to release their names:

Dr. Robie Macdonald, FRSC, (*Peer Review Monitor)* Fisheries and Oceans Canada , Victoria, British Columbia

Dr. Ronald Atlas, Department of Biology, University of Louisville, Louisville, Kentucky

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Dr. Jacqueline Michel, Research Planning, Inc., Columbia, South Carolina

Dr. Jeffrey Short, JWS Consulting, Juneau, Alaska

The peer reviewers provided many very constructive comments, but they were not asked to endorse the conclusions or recommendations, nor did they see the final draft of the report before it was released. The identification of reviewers is done to recognize them for their valuable contributions and does not imply that they endorse the report or agree with its content. Responsibility for the final content of this report rests entirely with the Panel.

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LIST OF ACRONYMS

ACR: acute-chronic ratio
ADEC: Alaska's Department of Environmental Conservation
AER: Alberta Energy Regulator; Alberta Government agency
AhR: aryl hydrocarbon receptor.
AMOP: Arctic and Marine Oilspill Program; an international technical forum held in Canada since 1978
ANS: Alaska North Slope crude oil
API: American Petroleum Institute
AMRA: Arctic Marine Risk Assessment
ASMB: Alberta Sweet Mixed Blend
ASTM: formerly the American Society for Testing and Materials; an international organization that publishes consensus technical standards and analytical methods
AUV: autonomous underwater vehicle
BaP: Benzo(a)pyrene, a 5-ringed PAH
BAT: best available technology
bbl: barrel; a unit of volume that varies between countries and for different products. A barrel of crude oil is equivalent to 159 litres.
BSEE: Bureau of Safety and Environmental Enforcement, U.S. Department of the Interior
BOEM: Bureau of Ocean Energy Management, U.S. Department of the Interior
BSD: blue sac disease
BTEX: benzene, toluene, ethylbenzene and xylene isomers: monoaromatic hydrocarbons commonly present in petroleum, particularly in light crude oils and diluents
CAPP: Canadian Association of Petroleum Producers
CEPA: Canadian Energy Pipeline Association
CEWAF: chemically-enhanced water-accommodated fraction of oil
CNRL: Canadian Natural Resources Ltd.; an oil sands mining company
CROSERF: Chemical Response to Oil Spills Ecological Effects Research Forum
DFO: Department of Fisheries and Oceans (Canada)
DWH: Deepwater Horizon; subsurface blowout of very light crude oil in the Gulf of Mexico in 2010
EBSA: Ecologically and Biologically Significant Areas
EC50: median effective concentration
ECRC-SIMEC: Eastern Canada Response Corporation, certified by Transport Canada to respond to oil spills from tankers and oil-handling facilities in the Great Lakes, Quebec and Atlantic regions.
EDCF: effects-driven-chemical-fractionation
EEM: environmental effects monitoring
EEPP: Energy East Pipeline Project
EIA: environmental impact assessment
ERA: ecological risk assessment
ERCB: Energy Resources Conservation Board (Alberta), now replaced by AER
ESI: environmental sensitivity index
EVOS: *Exxon Valdez* Oil Spill; spill of Alaska North Slope crude oil into Prince William Sound in 1989
FID: flame ionization detector
GC: gas chromatography
HEWAF: high energy water-accommodated fraction of oil

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HFO: heavy fuel oil, e.g., Bunker C
HLB: Hydrophilic-lipophilic balance
HMW: high molecular weight
HPLC: high performance liquid chromatography
IFO: intermediate fuel oil, e.g., IFO 180
ISB: *in situ* burning (of oil on water)
ITOPF: International Tanker Owners Pollution Federation; a not-for-profit marine ship pollution response advisory body
K_{ow}: octanol-water partition coefficient
LC50: median lethal concentration
LMW: low molecular weight
MC-252: Macondo crude oil
MESA: Medium South American crude oil
MoA: Mode of action
MOA: Mechanism of action
MMW: medium molecular weight
MNA: monitored natural attenuation
MS: mass spectrometry
NAS: National Academy of Sciences (United States)
NAs: naphthenic acids
NEB: National Energy Board (Canada)
NEBA: net environmental benefits analysis
NGP: Northern Gateway Pipeline
NOAA: National Oceanographic and Atmospheric Administration, U.S. Department of the Interior
NRC: National Research Council (in both Canada and the United States)
NTSB: National Transportation Safety Board (United States)
OHCB: obligate hydrocarbonoclastic bacteria
OMA: oil-mineral aggregate
OPA: oil-particle aggregate
PAH: polycyclic aromatic hydrocarbon
PBCO: Prudhoe Bay Crude Oil
PHMSA: U.S. Pipeline and Hazardous Materials Safety Administration
ppm: parts-per-million; a unit of concentration; e.g., mg/L or mg/kg.
ppt: parts-per-thousand; a unit of concentration; e.g., g/L or g/kg.
PTO: pole treating oil
RO: response organization
ROS: reactive oxygen species
ROV: remotely operated vehicle
SAGD: steam-assisted gravity drainage
SAR: synthetic aperture radar
SARA: an acronym for the four major chemical classes of petroleum constituents: saturates, aromatics, resins and asphaltenes
SCAT: shoreline cleanup and assessment technique
SCO: synthetic crude oil

SSD: species sensitivity distributions
TAN: total acid number
TCRA: Transport Canada Risk Assessment
TLM: target lipid model
TMPEP: Trans Mountain Pipeline Expansion Project
TPAH: total polycyclic aromatic hydrocarbon
TPH: total petroleum hydrocarbon
TSB: Transportation Safety Board of Canada
TSPS: Tanker Safety Panel Secretariat, Transport Canada
TU: toxic unit
UCM: unresolved complex mixture
UME: unusual mortality event
VEC: valued ecosystem component
U.S. EPA: United States Environmental Protection Agency
USGS: U.S. Geological Survey
VOCs: volatile organic compounds
WAF: water-accommodated fraction of oil
WCMRC: Western Canada Marine Response Corporation
WSF: water-soluble fraction of oil

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GLOSSARY OF TERMS

Acute-chronic ratio: the ratio of the acute and chronic toxicity values for a given compound, usually the average of the ratios for a variety of species; used to estimate the chronic toxicity of a compound or a mixture of compounds, from the measured or modeled acute toxicity when no chronic toxicity data are available.

Alberta Sweet Mixed Blend: a major blend of light crude oil exported from Alberta; also used by Environment Canada as a reference for laboratory tests.

Annual Exceedance Probability: the chance or probability of a natural hazard event (usually a rainfall or flooding event) occurring annually and is usually expressed as a percentage. Bigger rainfall events occur (are exceeded) less often and will therefore have a lesser annual probability.

Aromatics: class of hydrocarbons comprising one or more benzene ring structures having alternating double bonds; may have one or more alkyl side chains in various positions, but no heteroatoms.

Aryl hydrocarbon receptor: a cellular protein that binds planar or plate-like polycyclic aromatic compounds with a shape and dimensions that resemble 2,3,7,8 tetrachorodibenzo(p)dioxin.

Asphaltenes: class of petroleum compounds of high molecular weight and complexity; defined as the oil fraction that precipitates in low molecular weight *n*-alkanes (e.g., C5-C7) but is soluble in toluene.

Bioaccumulation (or bioconcentration): the tendency of substances to accumulate in the body of organisms; the net uptake from their diet, respiration or transfer across skin and loss due to excretion or metabolism. The bioaccumulation factor (BAF) or bioconcentration factor (BCF) is the ratio of concentrations in tissue to concentrations in a source, i.e., water or diet.

Bioavailability (or biological availability): compound that is in a physical or chemical form that can be assimilated by a living organism; also, the proportion of a chemical in an environmental compartment (e.g., water) that can be taken up by an organism.

Biodegradation: a natural process of microbial transformation of chemicals, such as oil under aerobic or anaerobic conditions; oil biodegradation usually requires nutrients, such as nitrogen and phosphorus; transformation may be complete, producing water, carbon dioxide and/or methane, or incomplete, producing partially-oxidized chemicals.

Biomagnification: a food chain or food web phenomenon whereby a substance or element increases in concentration at successive trophic levels; occurs when a substance is persistent and is accumulated from the diet faster than it is lost due to excretion or metabolism.

Biomarkers: a term used in two different ways, depending upon discipline. In petroleum chemistry, a biomarker is a relic chemical relating its presence to the original biological source (microbial, plant or animal); biomarkers are usually poorly or non-biodegradable and so persist in the oil, enabling their use as internal standards in petroleum analysis. In environmental toxicology, a biomarker is a biochemical process, product or cellular response that indicates the organism's exposure to a pollutant and/or the toxic effects of the pollutant.

Bioremediation: an intervention strategy to enhance biodegradation of spilled oil (or other contaminants), ranging from no remedial action other than monitoring (**natural attenuation**) to nutrient addition (**biostimulation**) to inoculation with competent microbial communities (**bioaugmentation**).

Bitumen: heaviest class of petroleum, having high viscosity and density; widely considered to represent the residue from lighter oils that have undergone biodegradation over geological time.

Blue sac disease: a disease syndrome of fish embryos caused by a variety of environmental stresses,

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including exposure to PAH and to alkyl PAH.

C_n: a molecule, such as a hydrocarbon, having *n* carbon atoms.

Chemically-enhanced water-accommodated fraction of oil (CEWAF): a solution of hydrocarbons and a suspension of oil droplets created when a chemical dispersant is added to oil and water with stirring. See also WAF and HEWAF.

Conventional crude oil: commonly defined as liquid petroleum that flows in the reservoir and in pipelines and is recovered from traditional oil wells using established methods, including primary recovery and water flooding (e.g., condensates, light and medium crude oils), versus unconventional crude oils.

Crude oil: synonymous with petroleum; a naturally-occurring and typically liquid complex mixture of thousands of different hydrocarbon and non-hydrocarbon molecules.

CYP1A: a member of the cytochrome P_{450} family of proteins; an enzyme of vertebrates, including fish, birds and mammals, that catalyzes the addition of oxygen to double bonds as a first step in the metabolism and excretion of PAHs.

cyp1a: the gene that codes for CYP1A proteins.

Dilbit: bitumen diluted with a lighter petroleum class, such as condensate or naphtha, typically 70% bitumen and 30% diluent; also see Synbit.

Dispersion: suspension of oil droplets in water accomplished by natural wind and wave action, production of biological materials (biosurfactants) and/or chemical dispersant formulations.

Dispersant: a chemical or mixture of chemicals applied, for example, to an oil spill to disperse the oil phase into small droplets in the water phase.

EC50: the concentration of a substance that causes sublethal effects on the median or 50th percentile organism tested within a specified exposure time (e.g., 14 d EC50)

Effects-driven-chemical-fractionation (EDCF): the step-wise fractionation, toxicity testing and chemical analysis of complex mixtures of compounds to isolate and identify the constituents responsible for the toxic effects of the whole mixture.

Ecological risk assessment: process for analyzing and evaluating the possibility of adverse ecological effects caused by environmental pollutants.

Emulsification: formation of water droplets in an oil matrix (water-in-oil) or conversely oil droplets in a water matrix (oil-in-water) achieved by the action of agitation, such as wind and wave activity; can be unstable, separating into oil and water phases again soon after formation, or stable for months or years (e.g., 'chocolate mousse', a water-in-oil emulsion).

Environmental impact assessment: the process of measuring or estimating the environmental effects of pollutants, such as oil spills, relative to conditions at a reference site or to a time prior to a spill.

Evaporation: the physical loss of low molecular weight components of an oil to the atmosphere by volatilization.

Flame ionization detector (FID): used with analytical instruments like gas chromatographs to detect components of petroleum by combustion ionization, hence GC-FID

Fracking: see hydraulic fracturing.

Gas chromatography (GC): an analytical method used to characterize petroleum components; GC is combined with different detection methods, hence GC-FID, GC-MS, etc.

HEWAF: A solution of hydrocarbons and a suspension of oil droplets created when oil and water are

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mixed by high energy agitation. See also WAF and CEWAF.

Heteroatom: in petroleum, an atom such as nitrogen, sulfur and/or oxygen that is part of a hydrocarbon skeleton, such as found in the Resins fraction of crude oils

High molecular weight (HMW): relative term referring to the molecular mass of chemicals; in oil, asphaltenes would be typical of HMW compounds.

High performance liquid chromatography (HPLC): an analytical method for separating chemicals in solution.

Hydrocarbon: a chemical that is composed of only carbon and hydrogen; chemicals containing heteroatoms, such as nitrogen, sulfur and/or oxygen, are not hydrocarbons, even though they may petroleum constituents.

Hydraulic fracturing: also known as fracking'; an unconventional method for recovering liquid petroleum from shale oil deposits that otherwise would not release the gas and/or fluid in the reservoir.

Hydrogen sulfide (H_2S): a toxic, flammable and corrosive gas sometimes associated with petroleum.

Hydrophilic-lipophilic balance (HLB): a measure of the properties of dispersants and surface-active agents at the interface of polar and non-polar liquids, such as oil and water. The higher the surfactant HLB value, the more hydrophilic (water-loving) it is.

Hyporheic flow: the perculating flow of water through the sand, gravel, sediment and other permeable soils under and beside the streambed.

Intermediate fuel oil: the heaviest commercial class of refined petroleum diluted with a lighter refined product, commonly burned in furnaces, boilers or ship engines; e.g., IFO 180.

Isomers: chemicals that have the same molecular formula (i.e., elemental composition) but different structures; may also have different properties, including water solubility, biodegradability and toxicity.

K_{ow}: the partition coefficient describing the equilibrium concentration ratio of a dissolved chemical in octanol versus in water, in a two-phase system at a specific temperature; used in prediction of toxicity.

LC50: the concentration of a substance toxic to the median or 50th percentile organism tested within a specified exposure time (e.g., 96 h LC50).

Lacustrine: relating to lakes.

Low molecular weight: relative terms referring to the molecular mass of chemicals; in oil, monoaromatics and aliphatics up to C_{10} would be typical of these compounds.

Mass spectrometry: an analytical method used for detailed characterization of petroleum components, often in combination with GC, hence GC-MS.

Mechanism of action (MOA): describes a functional or anatomical change at the molecular level.

Medium molecular weight: relative terms referring to the molecular mass of chemicals; in oil, 3 to 6-ringed PAH and aliphatics up to C_{20} would be typical of MMW compounds.

Mineralization: complete oxidation of a compound (e.g., hydrocarbon) to carbon dioxide and water; may be accomplished by a single species of organism or by a community of microbes.

Mode of action (MoA): describes a functional or anatomical change at the cellular level, resulting from the exposure of a living organism to a substance.

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Monitored natural attenuation: a remediation strategy in which there is no intervention but the site is monitored using various parameters.

Monoaromatics: aromatic hydrocarbons having only a single benzene ring; may also have one or more alkyl side chains.

Naphthenic acids: a class of polar petroleum compounds that contributes to aquatic toxicity and to petroleum infrastructure corrosion by contributing to TAN.

Natural attenuation: remediation of a contaminated site by natural processes alone, without human intervention; also see monitored natural attenuation.

Obligate hydrocarbonoclastic bacteria: bacteria that have adapted to use hydrocarbons as their sole source of carbon and energy for growth and metabolism.

Oil-mineral aggregate (OMA): floc containing oil adhering to mineral particle, which may float, sink or re-suspend in a water column; a process initially referred to as clay-oil flocculation.

Oil-particle aggregate (OPA): more general term than OMA, describing oil adhering to particles that may be inorganic (minerals) and/or organic, including microbial cells.

Partitioning: the diffusion of compounds between two immiscible liquid phases, including water and oil droplets and water and lipid membranes.

Petroleum: synonymous with crude oil; a naturally-occurring complex mixture of thousands of different hydrocarbon and non-hydrocarbon molecules.

Photooxidation: oxidation due to the influence of photic energy, usually from UV light.

Photo-enhanced toxicity: increased toxicity due to photooxidation *in vivo.*

Polycyclic aromatic hydrocarbons: (PAHs) a subclass of aromatic hydrocarbons having two or more fused benzene rings; may also have one or more alkyl side chains, generating large suites of isomers; some are considered 'priority pollutants' because of their toxicity and/or potential carcinogenicity.

Reactive oxygen species (ROS): a group of chemicals that can cause cellular damage due to their high reactivity; are released during reactions that add oxygen to double bonds. In turn, ROS can react with double bonds in lipids, proteins and nucleic acids to change their structure and function.

Resins: a solubility class of poorly characterized, polar petroleum compounds in which each molecule contains one or more atoms of nitrogen, sulphur and/or oxygen in a hydrocarbon skeleton.

Riparian: related to being situated on or dwelling on the bank of a river or other body of water, such as a lake or tidewater.

Riverine: relating to lakes.

SAGD (steam-assisted gravity drainage): an unconventional *in situ* method to produce bitumen from deep oil sands deposits without surface mining operations.

Saturates: class of hydrocarbons that may be straight-chain, branched-chain or cyclic, in which all carbon atoms have single bonds to either carbon or hydrogen.

Shale oil: also known as 'tight oil' (but not to be confused with 'oil shale'); liquid petroleum that is produced from shale oil reservoirs, typically by hydraulic fracturing methods.

Sour crude: petroleum that has a >1% total sulfur content that may be present as hydrogen sulfide and/or as organic forms of sulfur in a hydrocarbon backbone.

Species sensitivity distributions (SSD): compare the cumulative proportion of species (percentile)

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affected by a chemical to a toxicity endpoint measured for each species (e.g., 96 h LC50); SSD model assumes that species sensitivity is randomly distributed.

Sweet crude: petroleum with low total sulphur content, variously defined as <0.5% or <1% sulphur.

Synbit: bitumen diluted with synthetic crude oil, typically at 1:1 ratio.

Synthetic crude oil: a partially-refined fraction of bitumen; may be used as a diluent to make dilbit for transport.

Target lipid model: estimates the aqueous concentration of organic compounds, or mixtures of organic compounds, that cause toxicity by narcosis.

Total acid number (TAN): a measure of the acidity determined by the amount of potassium hydroxide in milligrams that is needed to neutralize the acids (typically naphthenic acids) in one gram of oil; used by refineries as an indicator of potential corrosion and scale production.

Total petroleum hydrocarbons (TPHs): the total mass of all hydrocarbons in an oil or environmental sample, including the volatile and extractable (non-volatile) hydrocarbons; may be further defined by stating the analytical method used, e.g., GC-detectable TPH or TPH-F (TPH measured by fluorescence), which vary in their rigour.

Total polycyclic aromatic hydrocarbons (TPAHs): including alkyl-PAHs and parent (unsubstituted) PAHs; the sum of all concentrations of PAHs measured by GC-MS.

Toxic unit (TU): ratio between the concentration of a compound in water and a toxic endpoint (e.g., 96 h LC50).

Unconventional crude oils: petroleum that does not flow readily in the reservoir and/or must be produced by using unconventional methods, such as surface mining of shallow bitumen deposits, steam-assisted gravity drainage (SAGD) for *in situ* extraction of deep bitumen deposits, cyclic steam injection for heavy oils, or horizontal drilling with hydraulic fracturing for recovery of light shale oils.

Unresolved complex mixture (UCM): petroleum constituents that are not resolved by conventional GC and appear as a 'hump' in the gas chromatogram; comprises many hundreds or thousands of unresolved isomers.

Unusual mortality event (UME): term coined by NOAA to describe a greater-than-usual rate of mortality of marine mammals.

Volatile organic compounds (VOCs): chemicals having high vapour pressure at room temperature (and corresponding low boiling point) that therefore tend to evaporate or sublimate into the air; for example, BTEX.

Water-accommodated fraction of oil (WAF): hydrocarbons that will partition from oil to water during gentle stirring or mixing; may contain droplets, in contrast to water-soluble fractions (WSF).

Water-soluble fraction of oil (WSF): aqueous solution of hydrocarbons that partition from oil; does not include droplet or particulate oil. See also CEWAF and HEWAF.

Weathering: a suite of changes in spilled oil composition and properties brought about by a variety of environmental processes including spreading, evaporation, photooxidation, dissolution, emulsification and biodegradation, among others.

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EXECUTIVE SUMMARY

A panel of leading experts on oil chemistry, behaviour and toxicity reviewed the current science relevant to potential oil spills into Canadian marine waters, lakes, waterways and wetlands. The review, which examined spill impacts and oil spill responses for the full spectrum of crude oil types (including bitumen, diluted bitumen and other unconventional oils), is among the most comprehensive of its kind. It surveyed scientific literature, key reports and selected oil spill case studies, including tanker spills, an ocean rig blowout, pipeline spills and train derailments. The Panel also consulted industry, government and environmental stakeholders across the country.

The Panel found that the dozens of crude oil types transported in Canada exist along a chemical continuum, from light oils to bitumen and heavy fuels, and the unique properties of each of these oil types (their chemical 'fingerprints') determine how readily spilled oil spreads, sinks, disperses, impacts aquatic organisms, including wildlife, and what proportion ultimately degrades in the environment. Despite the importance of oil type, the Panel concluded that the overall impact of an oil spill, including the effectiveness of an oil spill response, depends mainly on the environment and conditions (weather, waves, etc.) where the spill takes place and the time lost before remedial operations.

The Panel recommends that this critical research should concentrate on seven general high-priority research needs:

High-Priority Research Needs Identified by the Expert Panel

1. *Research is needed to better understand the environmental impact of spilled crude oil in high-risk and poorly understood areas, such as Arctic waters, the deep ocean and shores or inland rivers and wetlands.*
2. *Research is needed to increase the understanding of effects of oil spills on aquatic life and wildlife at the population, community and ecosystem levels.*
3. *A national, priority-directed program of baseline research and monitoring is needed to develop an understanding of the environmental and ecological characteristics of areas that may be affected by oil spills in the future and to identify any unique sensitivity to oil effects.*
4. *A program of controlled field research is needed to better understand spill behaviour and effects across a spectrum of crude oil types in different ecosystems and conditions.*
5. *Research is needed to investigate the efficacy of spill responses and to take full advantage of 'spills of opportunity'.*
6. *Research is needed to improve spill prevention and develop/apply response decision support systems to ensure sound response decisions and effectiveness.*
7. *Research is needed to update and refine risk assessment protocols for oil spills in Canada.*

Timeframes for conduct of the recommended studies are provided by the Panel. This executive summary highlights these central conclusions and other priority research questions identified in the report chapters.

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What happens when crude oil spills into oceans, into lakes or into the waterways that wind through our forests, fields and towns? Canada produces some three million barrels of oil every day, importing hundreds of thousands more, and all of it travels somewhere. In this lake-and-river-rich country with the world's longest coastline, crossing water is inevitable when oil is transported. Whether in vast tankers traversing the sea or in pipelines, trucks and trains passing countless rivers, lakes and wetlands, oil is on the move. Some is drilled directly from the seabed where offshore rigs perch above ocean waves.

Meanwhile, accidents happen. Headline-grabbing calamities, such as the Deepwater Horizon blowout in the Gulf of Mexico in 2010, the *Exxon Valdez* spill off Alaska in 1989 and the *Arrow* spill off the coast of Nova Scotia in 1970, are periodic reminders that oil spills can shock the environment, the economy and the communities affected by them—at least in the short-term. Water is fouled. Wildlife is tarred. Fisheries and other industries struggle to recover.

The good news is that transporting oil at sea is safer than it has ever been. According to the International Tanker Owners Pollution Federation, large tanker spills occurred almost 14 times more often during the 1970s on average than they do today. Undersea blowouts during oil production and exploration are also rare (although Canadian offshore exploration and drilling is expected to increase). Less known is how much oil spilled from pipelines, trains and trucks reaches our lakes, rivers and wetlands (where oil can become trapped and remain concentrated causing more harm or creating more concern because towns and cities are nearby). However, while big oil spills from grounded tankers, oil rigs, pipeline ruptures or train wrecks are guaranteed newsmakers, in truth most of the oil-related chemicals that make it into our oceans arrive from natural seepage, routine tanker maintenance and runoff from land.

Even so, the potential impact of spills into Canadian waters during the transport of oil can be profound.

The Royal Society of Canada Expert Panel report addresses this impact. Its purpose is to better understand what we know and, perhaps more importantly, what we need to find out. Chief among the report's aims is to provide a roadmap to research questions concerning how crude oils, including diluted bitumen and other unconventional oils, behave and how they affect ecosystems and communities after spilling into the changeable and weather-affected environments of Canada's vast marine and inland waters.

The Expert Panel has highlighted hundreds of conclusions and identified a long list of research needs in its extensive report. This executive summary highlights only the most pressing of these research priorities. Answers to these research questions are considered by the Panel to be essential for equipping policy makers, oil industry decision makers, oil spill responders and other Canadians with critical tools to better anticipate spills and their consequences and to better protect Canada's marine and inland waters from the adverse effects of spilled oil. The Panel has consolidated seven general, high-priority research needs (see the box above) in this executive summary from hundreds of research priorities identified in the report chapters. The rationale for these research needs, and more detailed descriptions, can be found within the report itself.

CHAPTER 1: INTRODUCTION

The Royal Society of Canada (RSC) Expert Panel was established in response to a request from the Canadian Energy Pipeline Association (CEPA) and the Canadian Association of Petroleum Producers (CAPP). The request was the result of a widespread recognition within industry, governments and elsewhere that Canadians should know what to expect in the event of an accidental spill, and that those who move oil and respond to spills have the information they need to protect our environment, economy and communities across the country.

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The Panel, composed of international experts on oil chemistry, behaviour and toxicity, reviewed the current science relevant to crude oils spilled into Canadian marine waters, lakes, waterways and wetlands. (Spills of gasoline, diesel and other refined fuels were not considered.) The Panel relied on scientific literature, key reports and selected oil spill case studies, including well-known tanker spills (e.g., the *Arrow* spill in 1970 and the *Exxon Valdez* spill in 1989), the Deepwater Horizon spill of 2010, pipeline spill ruptures and train derailments.

The Panel's work also involved extensive consultations with key industry, government and environmental stakeholders across the country. These included representatives from CEPA and CAPP, government agencies in Canada (Environment Canada, Natural Resources Canada, Fisheries and Oceans Canada, Alberta Innovates) and the United States (the National Oceanographic and Atmospheric Administration), private sector consultants, oil spill response agencies (e.g., the Eastern Canada Response Corporation), non-government organizations (e.g., Greenpeace), as well as other academics and interested individuals. Formal consultations included public forums involving open, online access held in Calgary in February 2015 and in Halifax in April 2015. A third Panel meeting in June 2015 included informal discussions with attendees at the 38th international Arctic and Marine Oilspill Program (AMOP) technical conference in Vancouver.

These consultations and the scientific review examined spill impacts and oil spill responses for the full spectrum of oil types, from ultra-light condensates and light oils to bitumen, diluted bitumen and heavy fuels. Many of the largest knowledge gaps were found to be associated with the chemical composition and environmental behaviour of emerging petroleum types, including diluted bitumens and other unconventional oils.

CHAPTER 2: CHEMICAL COMPOSITION, PROPERTIES AND BEHAVIOUR OF SPILLED OILS

While many Canadians think of the oil travelling by pipeline, train, truck or tanker as much the same, the crude oil crossing the country or plying its offshore waters each day represents dozens of different types. This may seem trivial to some, but if the oil spills into water, the type of oil involved can make a world of difference: How much damage it does, how easy it is to clean up and how readily the oil degrades in the environment.

Each of the oil types transported in Canada is a complex mixture of thousands of chemicals. While these different types can be thought as existing along a kind of chemical continuum (from ultra-light oil condensates and light oils to heavy crude oils and the thick bitumen commonly associated with Canada's oil sands), each has its unique 'chemical' fingerprint.

The Panel found that this chemical fingerprint is a key predictor of not only the physical properties of the oil (e.g., how heavy or thick it is), but also its behaviour in the environment (e.g., how it spreads, sinks or disperses in water), its toxic effects on aquatic organisms and humans, and its susceptibility to degradation by 'weathering' (i.e., changes to the oil caused by the sun, waves, weather conditions and microorganisms in the environment). How the fingerprint of each spilled oil type changes in the environment is an important tool for spill responders for monitoring cleanup efforts and setting cleanup goals.

Although the Panel found the chemical composition and behaviour of many oil types have been well-studied, more research is needed to better understand the chemistry, properties and spill behaviour of newer, less-familiar oils, such bitumen, diluted bitumen blends and other unconventional oils.

CHAPTER 3: EFFECT OF ENVIRONMENT ON THE FATE AND BEHAVIOUR OF OIL

The unique features of the environment where an oil spill takes place are at least as important as the type of oil in determining effects on aquatic ecosystems.

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Saltwater strait s, freshwater lakes, running rivers and dense wetlands are home to distinctive combinations of physical characteristics, water and sediment chemistry and natural communities of microorganisms that can transform oil as it spills and spreads. Microorganisms, for example, degrade various hydrocarbons found in different oil types to varying degrees, and their impact is often an important part of oil spill cleanup strategies. Sunlight, wind, waves and weather conditions can physically and chemically transform a spill. Temperature, dissolved oxygen, nutrient supply, salinity and pH also alter the composition and behavio ur of contaminating oil . These chan ges to the chemistry of oil are crucial factors affecting how spilled oil spreads, affects aquatic organisms and people or lingers in the environment.

Indeed, t he Panel found that, despite the importance of oil type, the overall impact of an oil spill, including the effectiveness of an oil spill response, depends mainly on the environmental characteristics , the conditions where the spill takes place and the speed of response

The impact on spilled oil of the characteristics and conditions of a spill site has been carefully studied in some environments, but knowledge gaps remain. In particular, research is needed to better understand what happens to oil spilled into the cold, icy, yet ecologically sensitive waters of the Arctic , where interest in oil exploration, production and shipping is on the rise. Similarly, little is known about the fate of oil and its impact in permafrost areas or in marine environments covered in ice. Microorganisms that break down oil are considered less active when temperatures are near freezing, but this relationship may not be as clear as we think and further study is needed.

High-Priority Research Need #1

Research is needed to better understand the environmental impact of spilled crude oil in high-risk and poorly understood areas, such as Arctic waters, the deep ocean and shores or inland rivers and wetlands.

i. Research is needed to assess the complex interactions among physical, chemical and biological factors unique to Arctic conditions (e.g., extreme cold temperatures, permafrost ecosystems, snow and ice) and different types of spilled crude oil. *(Timeframe: Within 5 years)*
ii. Research is needed to assess the fate and behaviour of oil spilled into freshwater ecosystems, especially in northern bogs, fens and areas of permafrost. *(Timeframe: Within 5 years)*
iii. Research is needed to evaluate risks associated with the shipment of fuel oil to communities in the Arctic. *(Timeframe: Within 5 years)*
iv. Research is needed to assess the risks of deep sea blowouts in the Beaufort Sea and in areas of the Atlantic coast that support commercial and subsistence fisheries, including research into the behaviour of oil on the surface with and without ice and the effects of subsurface oil plumes, residual oil deposited on deep sea sediments, oil stranded along shorelines and in backwater, marshy areas, and the impact of dispersant additions. *(Timeframe: 5-10 years)*
v. Research is needed to assess the risks of pipelines in Arctic freshwater environments, with an emphasis on the Mackenzie River. *(Timeframe: 5-10 years)*
vi. Research is needed to investigate the fate of unrecovered oil in rivers where it can interact with ice, substrates, woody debris, bed sediments, groundwater and engineered structures. *(Timeframe: 5-10 years)*

CHAPTER 4: OIL TOXICITY AND ECOLOGICAL EFFECTS

Oil spills can have significant consequences for aquatic ecosystems. These effects can be both short-lived and long-lasting. In the days following a spill, f loating oil smothers mollusks, plants and other species at the shoreline. Oil on birds and mammals destroys their thermal insulation and buoyancy. Some chemical

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components of spilled oil dissolve in water and kill fish and other aquatic creatures (before they typically break down quickly and disappear). Other chemicals, such as polycylic aromatic hydrocarbons (PAHs), can persist in the water and cause chronic health effects for aquatic species that show up months or years later.

Light oils contain more compounds that are acutely toxic to aquatic organisms than medium or heavy oils. On the other hand, heavy oils contain more of the chronically toxic alkyl PAHs. The Panel could not conclude that diluted bitumens present a greater or lesser health risk to most species than other oils because there are too few data available on toxicity. However, there may be greater risk to bottom - and sediment-dwelling organisms due to the tendency for diluted bitumens to sink in fresh water under certain conditions.

The characteristics of the oil spill location and its environment determine how spilled oil affects aquatic biota. Oil spills into fresh water, for example, are generally smaller than marine spills, but they may have a greater relative impact because the oil can't be diluted and degraded by the large volumes of water available at sea. Inland shorelines and sediments are more likely to become fouled, and less time is available to contain a freshwater oil spill before it contaminates sensitive habitats.

It is not only the spills themselves that threaten ecosystems, but oil spill cleanup can be damaging as well. Physical cleanup (e.g., removing oiled vegetation or tarred shoreline) destroys habitat and can cause erosion or the buildup of silt. Habitat damage reduces the abundance and productivity of native species and fosters invasive species. Using chemicals to disperse spilled oil often means surface oil is transferred to subsurface water at concentrations that can be toxic to aquatic life (especially to fish embryos). More research is needed on spill cleanup methods that limit habitat damage and the threats to wildlife.

Oil spill impacts on aquatic ecosystems are difficult to measure. In many cases, information about the ecology of a site before a spill occurs (i.e., baseline data) is scarce or missing altogether. Baseline monitoring is typically the responsibility of provincial and federal government departments as part of environmental and natural resource management. Coordination and collaboration is needed between the oil industry and these government agencies to ensure that monitoring addresses the needs for data to assess the distribution and effects of spilled oil in ecosystems most at risks of spills.

Assessment of oil spill impacts on 'ecosystem services' should be considered. Ecosystem services are the benefits provided by ecosystems to humans that contribute to making human life both possible and fulfilling. Further research is also required to better understand how the toxicity of spilled oil is affected by its interaction with the environment in which it was spilled.

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High-Priority Research Need #2

Research is needed to increase the understanding of effects of oil spills on aquatic organisms, populations, communities and ecosystems.

i. Research is needed to investigate the cumulative and interactive effects of co-exposure to oil and other human-induced and natural environmental stressors, such as industrial and municipal pollution, extreme temperatures, salinity, low oxygen concentrations and elevated concentrations of suspended sediments. *(Timeframe: 5-10 years)*
ii. Research is needed on the effects of spilled oil on populations and community structure of aquatic biota. *(Timeframe: 5-10 years)*
iii. Research is needed to understand the indirect effects of oil spills on ecological processes, such as interactions within and among trophic levels in aquatic food chains. *(Timeframe: 10+ years)*
iv. A program of research is needed on the resilience of aquatic ecosystems affected by oil spills, particularly at sites of past spills and in ecosystems unique to northern Canada (e.g., bogs, fens, etc.) at a high risk of oil exposure. *(Timeframe: 10+ years)*
v. Research is needed to investigate the socioeconomic impacts of oil spills as a first step in implementing an ecosystem services approach to oil spill impact assessments *(Timeframe: 10+ years)*

CHAPTER 5: MODELING OIL SPILLS IN WATER

Knowing what to expect and how to respond when oil spills into an ocean, lake or river is no mean feat. The complex chemistry of each oil type makes it difficult to predict how the oil will act and change when it meets the equally complex water chemistry, ecology and conditions at the particular site where a spill takes place. For this predictive work, scientific models are invaluable tools.

Scientific modeling creates conceptual or mathematical representations of complex real-world phenomena that can't be readily observed. Scientific models of oil spills use what we know from experiments, previous spills and other information to approximate what happens when oil of a particular type spills in particular circumstances. Scientists are constantly adding information and refining these models to improve their accuracy for predicting spill consequences and for understanding the best spill responses.

Early models of oil behaviour and transport relied heavily on experimental observations. Since the early 1980s, advances in oil spill modeling focused mainly on oil dispersion (the formation of oil droplets), the formation of oil particle aggregates , emulsification, evaporation and the general transport of oil in open water as well as in other types of ecosystems. More recently, researchers have developed more advanced numerical models of these various processes to better predict oil's behavio ur and changes in situations where no direct measurements can be made, such as in deep water or in the Arctic.

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The Panel found that while scientific modeling has made many advances in predicting how the environment can influence spilled oil and its behavio ur (through dispersion, biodegradation, dissolution, etc.), more research is needed to improve models of oil-in-ice effects, oil dispersion by waves, oil droplet formation from blowouts, the formation of oil particle a ggregates and the biodegradation of oil droplets under various environmental conditions (such as temperature, salinity, nutrient availability, light and chemical dispersants).

High-Priority Research Need #3

A national, priority-directed program of baseline research and monitoring is needed to develop an understanding of the ecological characteristics of areas that may be affected by oil spills in the future.

i. Research is needed to collect and evaluate baseline information from high-risk, poorly understood areas, such as the Arctic and other less-studied Canadian environments. *(Timeframe: Within 5 years)*
ii. Research is needed to understand the current status of sensitive and highly-valued species and vulnerable habitats for specific, pre-defined locations in Canada representing a range of human disturbance, from relatively undisturbed to highly disturbed. *(Timeframe: Within 5 years)*
iii. Research is needed to create ecosystem sensitivity maps, prioritized according to recent relative risk assessments, the intensity of current and potential future human use, the relative sensitivity of ecosystems and geographic gaps (e.g., in large areas of inland Canada). *(Timeframe: Within 5 years)*
iv. Research is needed to understand the natural variability of population and community metrics (e.g., abundance, diversity, productivity) across physical and chemical gradients as well as across time (seasonal and annual). *(Timeframe: Within 5 –10 years)*
v. Research is needed to identify other anthropogenic stressors that could influence the effects of oil spills. *(Timeframe: Within 5–10 years)*

CHAPTER 6: A REVIEW OF SPILL RESPONSE OPTIONS

Just as types of crude oil are far from uniform and the environments and conditions where spills occur are many, effectively responding to oil spills is complicated. Decisions about what response is best and the likelihood of success depend not only on the oil type, environment and weather conditions, but also on technical and logistical factors (such as the responders' knowledge and skills, the availability of personnel and equipment, time constraints, regulatory approvals, health and safety criteria, etc.), as well as financial concerns (such as the cost and economic impacts of the spill). Other considerations include the level of community engagement.

There are three main categories of oil spill responses. The first simply relies on natural processes to disperse and degrade spilled oil. For instance, naturally occurring microorganisms can remove or break down some of the hydrocarbons and other chemicals in the oil (called 'natural attenuation'). Evaporation can also help remove volatile and lighter weight components of spilled oil, while exposure to sunlight and oxygen causes the natural photooxidation of some of the oil's aromatic compounds. The second type of response involves physically containing and removing spilled oil, often using booms and skimmers on the water or washing a nd scraping at shore. Thick slicks of oil can also be burned at a spill site. The third response type uses biological and ch emical methods. This can involve methodologies to enhance the growth of oil-degrading microbes and/or plants on contaminated sites (phytoremediation) or the application of chemical dispersants that break up oil slicks into small droplets that become diluted into the water column where they are eventually also biodegraded.

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Choosing the best response or combination of responses depends on the unique circumstances of each spill. Among these are weather, wave height, ice conditions, daylight and ecological factors, including the risk to fish, invertebrates and other wildlife. Technical and economic factors also play a role, as well as the inherent effectiveness of the response strategy being considered.

The Panel found that most of what is known about oil spill response technologies has been developed through laboratory work and case studies. A better understanding of appropriate spill responses in the Arctic and in snow and ice conditions is vital. The Panel recommends carefully controlled field studies to help close this knowledge gap (without significant negative impact on the environment). Research is also needed to better understand less familiar response methods, such as anaerobic biodegradation in sediments, and emerging technologies (e.g., bioventing, air sparging, etc.) for aiding in the cleanup of anaerobic or anoxic sea floor and lake bottom environments contaminated by sunken oil. What happens to chemically-dispersed oil in both the deep sea and on its surface also needs to be studied using controlled empirical experiments.

High-Priority Research Need #4

A program of controlled field research is needed to better understand spill behaviour and effects across a spectrum of crude oil types in different ecosystems and conditions.

i. Controlled field experiments on oil spills (sanctioned by the federal government through a new permitting system) with rigorous statistical designs are needed at a variety of sites representing different coastal marine and freshwater ecosystems and conditions. *(Timeframe: Within 5 years and beyond)*
ii. Research is needed at the site of previous oil spills in Canada to increase our understanding of the effects of spilled oil over the long-term and of the extent of natural cleanup. *(Timeframe: Within 5 years)*

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High-Priority Research Need #5

Research is needed to investigate the efficacy of spill responses and to take full advantage of 'spills of opportunity'.

i. Research is needed to help develop effective oil spill response measures tailored to the Arctic, including studies that explore the interactions of oil with permafrost and ice or that examine the microbial degradation of oil at low temperatures. *(Timeframe: Within 5 years and beyond)*
ii. Advanced planning and contingency funds are needed to support research on the fate, behaviour and effects of real-world oil spills as they occur ('spills of opportunity') in the short, medium and long-term, including studies of the relative effectiveness of response measures. *(Timeframe: Within 5 years)*
iii. Indigenous peoples from all parts of Canada need to be involved the development of research protocols, in oil spill preparedness, cleanup and remediation/restoration, including involvement in the investigations of 'spills of opportunity'. *(Timeframe: Within 5 years)*
iv. Research is needed (along with the engagement of Indigenous peoples and other stakeholders and economic analysis) to address the long-standing remediation question "how clean is clean?" *(Timeframe: 5-10 years)*
v. Research is needed to develop and improve methods for remediation, reclamation or restoration of damaged marine and freshwater habitats following oil spills. *(Timeframe: 5-10 years)*
vi. Research is needed on the efficacy and environmental impacts of conventional and new oil spill remediation options, particularly in Arctic and freshwater ecosystems. *(Timeframe: 5-10 years)*

CHAPTER 7: PREVENTION AND RESPONSE DECISION MAKING

The best way to protect aquatic environments from the sometimes devastating impacts of spilled oil is to prevent spills from happening in the first place . That is, reducing the likelihood of accidental spills is always more effective than managing the risks (to the environment or to the economy) after a spill has occurred. This principle is particularly true in the sensitive ecosystems where spills can cause catastrophic or irreversible consequences, such as in the Arctic where industrial activities (e.g., offshore oil and gas, mining), urban growth and climate-related changes to navigation routes are expected to increase tanker traffic in the years ahead.

All oil spill strategies emphasize prevention as the prior emergency management activity. Effective prevention combines an understanding of the science and technologies associated with oil operations and potential oil spills with a clear understanding of t he environment and conditions in which these activities are taking place. For example, properly designed pipelines or tankers can be built to withstand anticipated conditions (waves, wind, ice, etc.) that increase spill risk. Similarly , established procedu res, proper inspection and maintenance of equipment and training for extreme and adverse circumstances help reduce the chances of a spill.

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High-Priority Research Need #6

Research is needed to improve spill prevention and develop/apply response decision support systems to ensure sound response decisions and effectiveness.

i. A national guidance program for post-spill monitoring is needed to collect reliable, adequate, credible and consistent information on the fate and effects of oil in the environment. This program should be developed based upon consultations among industry, government, Indigenous organizations, and community stakeholders. (Timeframe: Within 5 years)
ii. Research is needed to develop methods to support the monitoring of oil-spill impacts and the fate of released oil. (Timeframe: 5-10 years)
iii. Research is needed to develop methods for the derivation of comprehensive mass balances for spilled and recovered oil. (Timeframe: 5-10 years)
iv. Research is needed to develop modeling methods to simulate and optimize individual and collective cleanup processes (e.g., booming, *in situ* burning, skimming, dispersion and bioremediation) for supporting response decision-making. (Timeframe: Within 5 years)
v. Research is urgently required on development and demonstration of oil spill response decision support systems, which can dynamically and interactively integrate monitoring and early warning, spill modeling, vulnerability/risk analysis, response process simulation/control, system optimization and visualization. (Timeframe: Within 5 years)
vi. Research investment is needed on trial tests and field validation of new prevention and decision-making methods to demonstrate feasibility, increase confidence for implementation and improve response capabilities. (Timeframe: Within 5 years)
vii. Research is needed to better quantify modeling uncertainties, evaluate their propagation and mitigate their impacts on spill response decision-making. (Timeframe: 5+ years)
viii. Further research and development are desired on environmental forensics, remote sensing and *in situ* measurement, early warning and diagnosis, and biological monitoring to improve spill prevention and decision-making. (Timeframe: 5+ years)
ix. Special attention of the above research should be given to some emerging issues (e.g., diluted bitumen, aging/subsea pipelines, railcars and the Arctic) to enhance effectiveness and confidence of prevention and response strategies and decisions. (Timeframe: Within 5 years)

Prevention policies and measures are often best informed by the timely monitoring and analysis of the causes and outcomes of spills when they do occur. Importantly, knowing the characteristics and ecological features of an environment before a spill occurs is central to understanding how it has been affected. Developing baseline data in areas where oil is transported should be an important priority for research.

Indeed, despite the best prevention efforts, spills happen, and then making sound and timely decisions about how to respond is a critical second line of defense. The Panel found that management strategies should be in place to identify the lead decision-making agencies in the case of a spill and to present clearly the steps to contain potential damage to human health, businesses and the environment. Decisions concerning what to do following an oil spill can occasionally mean weighing the potential benefits of a response against its possible harm or against the pros and cons of another approach altogether. Net environmental benefit analysis (NEBA) provides a helpful framework and has been widely used for supporting these decisions. The review also disclosed the limited research efforts in simulating, predicting and optimizing cleanup processes (e.g., *in situ* burning, skimming and dispersion) individually and collectively and evaluating their effects on response decisions. Inadequate decision support is one of the major challenges that limit the efficiency of current response practices. Due to limited attention and investment, existing decision support systems are rare and lack dynamic and interactive support from

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other modeling tools (risk analysis, spill modeling, NEBA, process simulation, etc.) and field validation. In addition, uncertainty is a major hindrance to improving efficiency and confidence of decision -making. These are especially true for the Arctic waters where the window of opportunity is significantly short. The Panel also noticed that advances in monitoring and information technologies such as remote sensing, geographic information systems, artificial intelligence and visualization have provided a set of cost -effective and powerful tools that can play a more important role in better addressing complexity and dynamics of spills and supporting sound response making and operations.

High-Priority Research Need #7

Research and work are needed to update, refine and focus risk assessments of oil spills in Canada.

i. Follow-up relative risk assessments are needed to build upon the Transport Canada assessments of marine spills, focusing on high-sensitivity areas. *(Timeframe: Within 5 years)*
ii. Research is needed to update and refine risk assessment methods to include such things as credible spill scenarios, analyses of seasonal differences in fate, transport and effects of oil (particularly for spills in winter) and the prediction of chronic toxicity. *(Timeframe: Within 5 years)*
iii. A comprehensive national database is needed to track the fate, behaviour and effects of various types of oil spilled and the efficacy of current and emerging oil spill countermeasures over a range of environmental conditions. *(Timeframe: 5-10 years)*
iv. Research is needed to expand species sensitivity distributions (SSDs) for acute and chronic toxicity of oil to aquatic biota. SSDs should be expressed as measured concentrations of total petroleum hydrocarbons and total polycyclic aromatic hydrocarbons. *(Timeframe: 5-10 years)*
v. Research is needed to extend models of chronic toxicity to a wider array of species and environmental (temperature, salinity, etc.) and life history variables. *(Timeframe: 10+ years)*

CHAPTER 8: RISKS FROM OIL SPILLS

Learning from history is important to understanding what's known about the risks posed by potential oil spills in Canada and, most significantly, what needs further study. The Panel reviewed the circumstances and outcomes for selected oil spill cases involving tanker accidents (e.g., the *Arrow* spill in 1970 and the *Exxon Valdez* spill in 1989), a major ocean-rig blowout (i.e., the Deepwater Horizon Gulf of Mexico spill in 2010), pipeline spills and train derailments that occurred in marine and fresh water in Canada and the United States over the past few decades.

Chief among the Panel's conclusions is that each case was unique in the combination of different physical, chemical and biological factors at the spill location, as well as in the cleanup and recovery measures used in the wake of each accident. These varied combinations of factors were critical for either increasing or decreasing the overall impact for each spill.

Delays in responding to the spilled oil affected the outcome of all case studies examined. Indeed, despite the obvious importance of weather, remote location and technological challenges facing each accident, human error (individuals and organizations) played a dominant role in affecting the impact of the spills across all case studies. Absent or inadequate planning, limited data analysis, inadequate training, poor communication, insufficient personnel and equipment, poor or no information sharing, and lapses in regulatory oversight were common to most, if not all, spill case studies.

From its case study review, the Panel found that the ability of aquatic ecosystems to recover from the shock of an oil spill may be influenced by the presence of other longer -term environmental stresses (e.g., habitat degradation caused by urban development, fishing pressure or water pollution from sewage

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discharges or agriculture). In most cases, the lack of pre-spill baseline data (i.e., information about the natural environment and ecology of each area) hampered efforts to predict or monitor the long-term effects of the oil spills on populations and communities of aquatic life. Similarly, monitoring following spills was not conducted according to any standard or consistent national protocols. The Panel's review of risk assessments of oil spills in Canada revealed a number of challenges, notably the lack of readily accessible data for use in the assessments and the need for increased sophistication of both exposure and effects analyses. In many cases, even if data were accessible, they were extremely limited, particularly for the Arctic and large portions of inland rivers, lakes and wetlands. The Panel found that the assumptions used in the risk assessments sometimes were overly optimistic given the experience gained from oil spill case studies. This was especially true for the spill response times assumed in the assessments.

CHAPTER 9: CONCLUSIONS

Crude oil spills are infrequent in Canada's coastal or inland waters. But the consequences of these spills into sensitive waterbodies can be profound. They can significantly affect not only the environment but also the economy of affected areas as well as human health and safety.

Canada's offshore oil and gas industry, meanwhile, is expected to grow. The production and transport of unconventional oils, such as diluted bitumens and Bakken crude oil, are likely to increase. Spills of these oils from offshore platforms, pipelines, tankers, rail, and other sources will continue to pose risks to Canadian aquatic environments and the communities that rely on them.

The Royal Society of Canada expert Panel prepared this report—based on a review of the science and consultations with key stakeholders—to better understand what is behind these risks. Among the Panel's many conclusions is a long list of research needs, including seven key research areas that should become top research priorities.

In particular, the Panel recommends that research needs to identify where most oil spills occur and why (e.g., pipeline spills into wetlands are more common than these spills into rivers; oil from truck spills are more likely to enter storm sewers before reaching rivers; etc.). Researchers need to examine past spill response records, current risk management processes and regulations to identify their weaknesses. Other critical knowledge gaps include developing a better understanding of environmental sensitivities that affect the impact of spilled oil. More research is also needed to understand how the type of oil, its source, the environment and the level of preparedness of spill responders combines to influence spilled oil's fate and effects.

These research gaps are significant. The data needed to assess oil spill risks in Canada are often either absent or widely scattered among government agency, industry and academic sources. Information needed to reliably assess the environmental sensitivity of areas at risk from oil spills is also very limited for large portions of Canada. Input of traditional knowledge from Indigenous peoples and other interested parties is needed.

Examples that demonstrate the need for more research are numerous. Many are documented in the Panel report. While scientific advances have significantly reduced the threat of oil spills in Canadian waters over the past few decades, much about the fate and effects of oil spills remains poorly understood.

To meet the research and oil spill-response needs identified in the expert report, the Panel recommends the conduct of coordinated multi-disciplinary research programs between industry, government and academia to further study the effects of oil spills on various marine and freshwater ecosystems, including wetlands. The program should also include Indigenous people for the provision of traditional knowledge and expertise. The science from these studies will provide a much needed database on the interaction and

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effects of spil led oil with its surrounding environment that will support science-based decision-making following future spill incidents to protect our aquatic environment and its living resources.

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CHAPTER 1: INTRODUCTION

Abstract

This chapter describes the rationale and mandate for a Royal Society of Canada Expert Panel established to provide a scientific review of our current knowledge and understanding of the behaviour and environmental impacts of a range of crude oils, including diluted bitumen, which may be accidentally released into Canadian marine or freshwater ecosystems.

To provide context for its evaluation of the risks associated with spills, an overview of global and Canadian information is provided on the probable causes and predicted frequency and size of crude oil spills into marine or fresh waters from exploration, production and transport operations. The global frequency of large spills (>700 tonnes) from oil tankers has decreased significantly in the past four decades and, fortunately, large spills due to tanker incidents in Canadian marine waters have been rare. Return periods (i.e., time between spills of a given volume) in Canada range between about 40 and 240 years (depending on spill volume); however, these return periods may not provide a reliable basis for predicting future occurrences in circumstances where the frequency and/or volumes of shipments are increasing. Diffuse sources, such as natural seeps and runoff from land-based sources, account for the majority of petroleum hydrocarbon inputs to oceans. Thus, while continued efforts to reduce tanker spills are necessary, attention should also focus on sources such as urban runoff and recreational boating because the spills are chronic and often occur in sensitive ecosystems. While freshwater spills associated with land-based extraction of crude oils (including the extraction of bitumen) and the transport of petroleum hydrocarbons (by pipeline, rail tankers and trucks) are expected to be at much smaller volumes than their marine counterparts, they may cause significant damage because of their higher probability to occur within populated areas and the proximity to water bodies lacking the dilution and dispersion capacity typically found within the marine environment.

Notwithstanding the apparent relative infrequency of crude oil spills into aqueous environments in Canada, the consequences of spills into sensitive marine and freshwater systems can be substantial, both with respect to impacts on human health, safety and the environment and with respect to economic impacts. Therefore, a primary focus of this Report is on the consequences of spills, no matter how low the probability may be. This focus is commensurate with the key questions posed by the sponsors and stakeholders.

1.1 Sponsors and Mandate of the Panel

The Expert Panel was established by the Royal Society of Canada (RSC) in response to a request from the Canadian Energy Pipeline Association (CEPA) and the Canadian Association of Petroleum Producers (CAPP) who recognized the need for a scientific review of our current knowledge and understanding of the behaviour and environmental impacts of a range of crude oils, including diluted bitumens, accidentally released into Canadian marine or freshwater ecosystems. The Panel was asked to address the following questions [with additional context added by the panel]:

1. How do the various types of crude oils compare in the way they behave when mixed with surface fresh, brackish or sea waters under a range of environmental conditions or when chemically treated for spill remediation?
2. How do the various crude oils compare in their [chemical composition] and toxicity to organisms in aquatic ecosystems?
3. What should be done to ensure optimal microbial activity for rapid remediation of aquatic systems impacted by a spill [and how do microbial processes affect crude oils in aquatic ecosystems, thereby modifying their physical and chemical properties, persistence and toxicity]?

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4. Is the research [and oil spill response] community able to relate, with reliable predictions, the chemical and physical [and biological] properties of crudes to their behaviour, [persistence], toxicity and ability to be remediated in water and sediments?
5. How should these scientific insights be used to inform optimal strategies and regulatory requirements for spill preparedness, spill response and environmental remediation?
6. Given the current state of the science, what are the priorities for research investments?

1.2 Scope of the Panel's Review

The Panel considered accidental releases of conventional and unconventional light, medium and heavy crude oils, including diluted bitumens. The Panel evaluated reports on the environmental behaviours of these oils and the effectiveness of current and emerging oil spill response models and technologies available for use in the event of accidental releases. Releases of biofuels or refined petroleum products (e.g., gasoline, diesel, aviation fuel, heating oil, etc.) were determined to be outside of the scope of this Report by the project stakeholders, although spills of Bunker C and IFO (intermediate fuel oil) were included for comparison.

> The terms 'conventional' and 'unconventional' oils are related to the techniques used to extract them from reservoirs, not to their chemical composition. Conventional oils include liquid crudes that flow in the reservoir and in pipelines. Such oils (e.g., condensates, light and medium crude oils) are recovered from traditional oil wells using established methods like primary recovery and waterflooding. Unconventional oils are produced using unconventional methods, such as surface mining of shallow oil sands, steam-assisted gravity drainage of bitumen (SAGD) and horizontal drilling with hydrofracturing ('fracking') for recovery of light shale oils ('tight oils').

The scope of this Report was intentionally focused on the accidental release of crude oils at the exploration and production source or as they are transported to refineries. Thus, the transport of crude oil by pipeline, ship, rail and truck was considered both with respect to the probability of accidental releases and consequences in the aqueous environment. Unplanned releases from offshore platforms, including subsurface blowouts, were included in light of the recent Deepwater Horizon (DWH) blowout and the expected growth of Canada's offshore oil and gas industry. The Panel focused on the Canadian environment; however, it reviewed and considered the applicability of case studies from the United States and other countries. Case studies were selected to represent different types of crude oil spilled into either marine or freshwater systems. Efforts were also made to find case studies applicable to cold climates.

This Expert Panel Report is not expected to provide exhaustive overview of previous oil spill research and case studies that have been cited in numerous existing reviews. While this information is clearly considered in the response to the above questions, emphasis has been placed on the expertise and experience of the individual Panel members to identify the most relevant and current literature in scientific journals and academic, government and consultant reports on current policies and practices in oil spill prevention and response, including the assessment of environmental impacts and remediation. The Panel recognizes the existence of numerous review articles related to recent oil spill events and the large amount of important research currently being conducted (e.g., research on diluted bitumen, and studies related to Natural Resources Damage Assessment following the DWH spill in the Gulf of Mexico). The conclusions and recommendations are based on a consideration of the large body of literature available up to the date of writing.

1.3 Consultation with Stakeholders

The Panel consulted with representatives of CEPA and CAPP, government agencies in Canada (Environment Canada, Natural Resources Canada, Fisheries and Oceans Canada, Alberta Innovates) and the U.S. (National Oceanographic and Atmospheric Administration [NOAA]), private sector consultants,

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first responders (e.g., Eastern Canada Response Corporation [ECRC -SIMEC]), non -government organizations (e.g., Greenpeace) , as well as individuals (e.g., academics). The formal consultations included public forums with open online access that were hosted by the Panel in February (Calgary, A B) and April (Halifax, NS) , 2015 that involved presentations , as well as question and answer sessions. A third Panel meeting in June , 2015 included informal discussions with attendees at t he 38th international Arctic and Marine Oilspill Program (AMOP) technical conference that included a dedicated session on diluted bitumens.

Several stakeholders emphasized that the Panel's work could be used as supporting information for government and industry policy and practice regarding spill prevention and emergency response. Some stakeholders were interested in receiving relevant information regarding transportation routing (e.g., to avoid or minimize risk to highly sensitive environments or specie s). Considerable interest was expressed in gaining an understanding of the relative risks associated with the transportation and handling of different types of crude oil. Some stakeholders wanted to know about the best available and emerging technologies for spill cleanup, as well as monitoring requirements to track the effectiveness of cleanup.

Stakeholders expected the Panel to take a comparative approach in order to better understand the relative impacts of conventional crude oil s and diluted bitumen. The results of the comparisons among crude oil types (and impacts in different marine and freshwater ecosystems) would then be useful when making decisions regarding preparedness and appropriate spill response procedures.

The stakeholders requested that the Panel clearly summarize what is known and not known about the key issues and make recommendations for actions and further research. Particular interest was expressed in the performance of different cleanup technologies and the long-term effects of spills.

1.4 Context for Oil Spills in Canada

The Panel delivers the following brief overview of information on the causes, frequency and size of crude oil spills into marine or fresh waters in Canada to provide context for its evaluation of the risks associated with spills.

1.4.1 Spills from Oil Tankers in Marine Waters

Although this report considers the full range of Canadian aquatic ecosystems, most of the research, response and media attention throughout the decades has focused on marine oil spills, particularly large - scale accidents involving oil tanker vessels. The largest spill due to an oil tanker accident was the *Atlantic Empress* in 1979 off the island of Tobago in the West Indies, where 287,000 tonnes of oil were released after a collision with another vessel, follow ed by fire and an explosion that sank the tanker. No impact studies were carried out, so it is not known what quantity of oil burned or sank, but only minor shore pollution was reported on nearby islands (ITOPF 2015a). The *Amoco Cadiz* grounding off the coast of Brittany in 1978 resulted in the release of 223,000 tonnes of light Iranian and Arabian crude oil and 4,000 tonnes of bunker fuel into heavy seas—causing the formation of an emulsion that increased the volume of pollutant by up to five-fold. Numerous shoreline types were affected and much of the oil became buried in sediments and entrapped in salt marshes and estuaries (ITOPF 2015b). The spill of 37,000 tonnes of Alaska North Slope crude oil into Prince William Sound, AK, from th e *Exxon Valdez* in 1989 was small in comparison, but led to the mortality of thousands of seabirds and marine mammals, a significant reduction in population of many intertidal and subtidal organisms, and reports of long-term environmental impacts (Spies et al. 1996).

Fortunately, large spills due to tanker incidents in Canadian marine waters have been rare. In terms of national involvement, one of the largest spills was in 1988 when the tanker *Odyssey* loaded with over 132,000 tonnes of North Sea Brent crude oil broke into two and sank 700 miles (~1,100 km) off the coast

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of Nova Scotia beyond Canada's Exclusive Economic Zone [1] (ITOPF 2015c). Fire started on the stern section as it sank and the surrounding oil caught fire. Due to rough weather, the Canadian Coast Guard was only able to come within about 2 km of the vessel while it was on fire. No shoreline effects occurred because of the distance of the spill from the coastline. Another well-known tanker accident off the Canadian coast occurred in 1970 when the *Arrow*, hauling 9,500 tonnes of Bunker C fuel oil from Aruba to Nova Scotia, encountered severe weather and gale force winds near Port Hawkesbury, NS, at the entrance to Chedabucto Bay. The tanker ran aground on Cerberus Rock spilling most of the fuel oil cargo and contaminating 75 miles of shoreline with residues of thick black sludge, which can still be found (Lee et al. 2003; Owens et al. 2008). Within the scientific community, the *Arrow* spill was unique because a section of the affected shoreline (Black Duck Cove) was intentionally left 'untreated' to enable scientific assessment of natural recovery processes.

The frequency of large spills (>700 tonnes) from oil tankers has decreased significantly in the past four decades. According to data from the International Tanker Owners Pollution Federation (ITOPF 2015d), an average of 1.8 large spills per year occurred internationally in the period 2010-2014 compared to an average of 24.5 large spills per year between 1970 and 1979. One large spill was recorded in 2014 - the sinking of a tanker in the South China Sea loaded with a cargo of about 3,000 tonnes of bitumen. The U.S. Department of the Interior's Bureau of Ocean Energy Management (BOEM) and Bureau of Safety and Environmental Enforcement (BSEE) produced a comprehensive summary report on the occurrence rates for offshore oil spills from U.S. Outer Continental Shelf (OCS) Platform and Pipeline Spill Data (1964 - 2010), Worldwide Tanker Spill Data (1974 - 2008), and Barge Spill Data for U.S. waters (1974 - 2008) (Anderson et al. 2012). According to this report, the rate of offshore oil spills in the U.S. has been declining since 1994, due mostly to major regulatory changes in the early 1990s (arising from the *Exxon Valdez* spill) that substantially eliminated the use of single-hull tankers by requiring double hulls or their equivalent. In 2013, a report by the Canadian Tanker Safety Panel Secretariat (TSPS) published two tables produced by GENIVAR, a professional services firm, showing the risk calculations for the probability and potential impacts of ship-source oil spills in Canada (Table 1.1) and internationally (Table 1.2). The significant figures in these two tables are from those reported by the data sources. The return periods (i.e., time between spills of a given volume) presented in these tables may not be a reliable basis for predicting future occurrences in circumstances where the frequency and/or volumes of shipments are increasing (e.g., along the southern British Columbia coastline if the proposed TransMountain Pipeline Expansion Project, with associated tanker traffic, is approved).

Table 1.1 Spill frequency estimates for ship-source spills occurring in Canada, calculated using only previous Canadian spill occurrences. *Data from TSPS (2013)*

	Return period, years [a]			
Spill volume, m^3	10-100	100-1,000	1,000-10,000	> 10,000
Crude oil	– [b]	–	–	–
Refined Cargo oil	1.7	10.0	–	–
Bunker oil	0.5	1.7	–	–
Total	0.4	1.4	–	–

[a] *estimated average number of years between spills*
[b] *probability could not be estimated due to the lack of spills in Canada in this category in the previous 10 years, indicating that the probability of a spill in this size range and category is remote.*

[1] *The exclusive economic zone (EEZ) is an area of the sea adjacent to and beyond the territorial sea, extending out to 200 nautical miles from the baseline. Within the EEZ, a coastal state has sovereign and jurisdictional rights over exploration and management (e.g., scientific research and protection of the marine environment) and economic exploitation of living and non-living resources in the waters above the seabed, in the seabed and beneath the seabed.*

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Table 1.2 Spill frequency estimates for ship -source oil spills in Canada, based on international and Canadian data. *Data from TSPS (2013)*

	Return period, years [a]			
Volume, m³	10-100	100-1,000	1,000-10,000	> 10,000
Crude oil	46.4	69.2	51.6	242.3
Refined Cargo oil	1.7	10.0	42.2	– [b]
Bunker oil	0.5	1.7	154.8	–
Total	0.4	1.4	20.2	242.3 [c]

[a] *estimated average number of years between spills*
[b] *probability could not be estimated due to the lack of spills in the previous 10 years in this category, indicating that the probability of a spill in this size range and category is remote*
[c] *i.e., a crude oil spill of >10,000 tonnes in Canadian waters is predicted to occur once every 242 years, based on the occurrence of only two such spills worldwide in the past 10 years, both from single -hulled vessels (the Tasman Spirit spill of ~30,000 tonnes in 2003 in Pakistan, and the Hebei Spirit spill of ~10,500 t onnes in 2007 in South Korea).*

In addition to the estimates above, ITOPF (2014) published actual oil tanker spill statistics, which show a declining trend in the number of tanker spills greater than 700 tonnes from 1970-2014 (**Figure 1.1**).

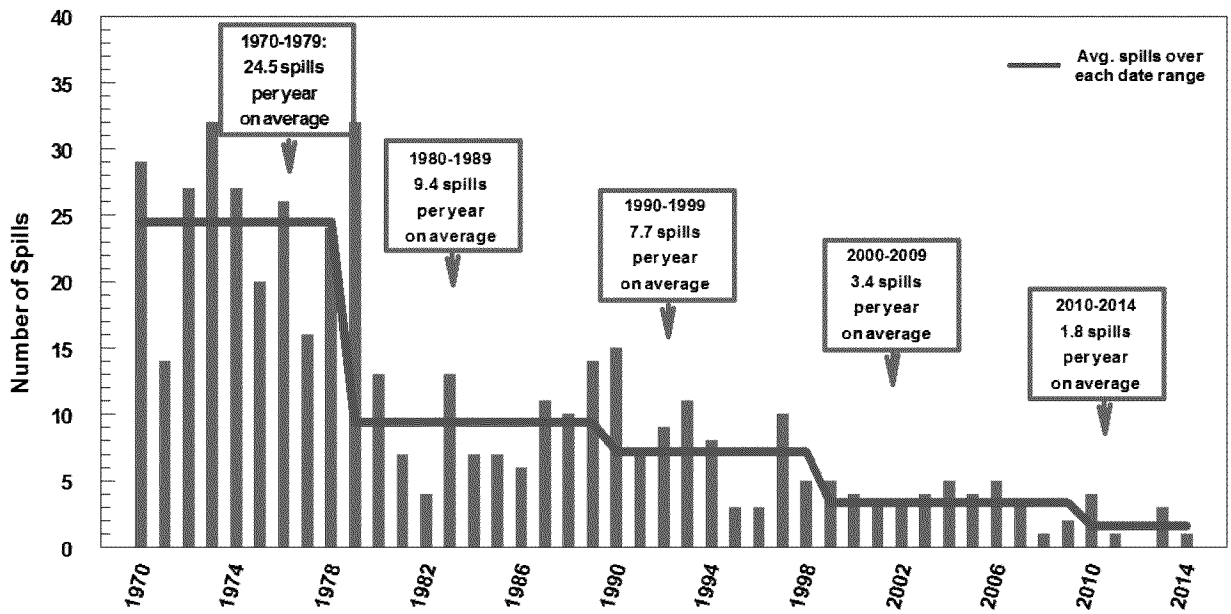

Figure 1.1 Number of large oil tanker spills (> 700 tonnes) occurring worldwide from 1970 to 2014 (grey bars), and average number of spills per decade (red line). *Adapted from ITOPF (2014)*

The 1989 *Exxon Valdez* incident increased public and political awareness about t he risks involved in the storage and transportation of oil and oil products. This awareness resulted in changes to regulations, including: the enactment of the 1990 *Oil Pollution Act* by the U.S. Congress, which regulates spills in the United States; regula tions within Canada under the *Canada Shipping Act* (2001) (http://laws-lois.justice.gc.ca/eng/acts/s-9/), which regulates tankers; and the *Canada Arctic Waters Pollution Prevention Act* (1985, most recently amended in 2014) (http://laws-lois.justice.gc.ca/eng/acts/A-12/), which includes regulations regarding the design of Arctic class ships.

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1.4.2 Accidental Releases from Offshore Oil and Gas Exploration and Production

Spills, blowouts and malfunctions may occur during any offshore oil and gas exploration activity. As defined by the U.S. NOAA, a blowout occurs when operators of a drilling rig are unable to control the flow of oil, gas or other fluids from the well, causing it to be released into the underground formation, marine environment and/or atmosphere. Until recently, the risk of blowouts was considered low in the oil exploration and development industry and thus of minor concern. For example, from 1979 to 1998, 19,821 wells were drilled within the Gulf of Mexico, with only 118 wells resulting in uncontrolled flows or blowouts - a 0.6% occurrence rate (API 2009). The reported blowouts were not considered a significant source of hydrocarbon releases to the environment due to effective blowout prevention (BOP) systems and the fact that they often sealed naturally and ceased flowing within a matter of hours or days. In 2004, the Canada Nova Scotia Offshore Petroleum Board (CNSOPB) predicted a 1-in-1800 chance per year of having any sort of deep water blowout off the continental shelf during exploratory drilling, with the probability of shallow water gas blowouts without a release of oil having a three - to four-fold higher probability of occurrence (Hurley and Ellis 2004). However, the DWH blowout in the Gulf of Mexico, which resulted in the continuous discharge of petroleum gas and crude oil into surrounding waters (4.2 million bbl [~600,000 m^3] over an 87 day period) and impacts on the Gulf shoreline and salt marshes, has changed the world's perception of the environmental risk associated with blowouts, with accompanying changes in regulatory requirements. For example, Canada's National Energy Board (NEB) is mandating same-season relief well capability or equivalent for the Arctic because the tracking and cleanup of oil spills under ice are major challenges in terms of response.

Only one major oil well blowout has occurred in Canadian offshore waters - the 1984 Uniake G-72 gas and condensate well off Sable Island approximately 150 nautical miles (~275 km) from Halifax, NS (Boudreau et al. 1999). In 2004, due to equipment failure, about 170 tonnes of crude oil was accidentally released from the Terra Nova production, storage and offloading platform on the Grand Banks, about 340 km east-southeast of St. John's, NL. The primary risk was to seabirds present in relatively high densities in the area of the spill. It was estimated that about 10,000 seabirds were killed by the spill (Wilhelm et al. 2007) as the oil slick was dispersed by natural physical processes.

Diffuse sources (natural seeps and runoff from land-based sources) account for the majority of petroleum hydrocarbon inputs to oceans (NRC 2003). The total input of petroleum into the sea worldwide from all sources over the period 1990-1999 was estimated by the NRC as approximately 1.3 million tonnes per year. Natural petroleum seeps and runoff from land-based sources contributed 46% and 11% of petroleum hydrocarbon inputs, respectively. Tank vessel spills contributed about 8% of total inputs, while operational discharges (e.g., bilge discharges and cargo washings) contributed about 3% of inputs. Furthermore, most spills from tank vessels are relatively small. Data from the ITOPF on oil tanker spills show that of nearly 10,000 incidents, 81% resulted in releases of less than 7 tonnes (ITOPF 2015d). More recent comparisons of relative contributions of petroleum hydrocarbons to the world's oceans confirm earlier conclusions (**Figure 1.2**).

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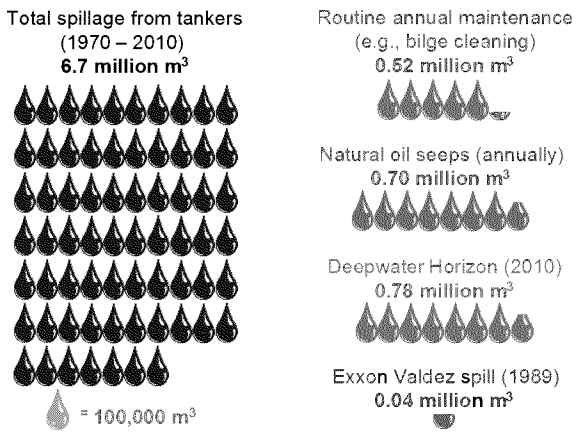

Figure 1.2 Volumes of oil released to marine waters from n atural sources and spills . *Data from NRC (2003); Mackenzie (2011)*

Examination of the proportion of oil released into North American coastal and offshore marine waters from natural and human activities reveals that natural seeps and land-based sources accounted for an even greater proportion of inputs (NRC 2003) (**Figure 1.3**). The NRC estimated total inputs to North American waters were 260,000 tonnes per year over the p eriod 1990-1999, with natural seeps contributing 61% of total inputs. Extraction of petroleum contributed about 1% of total inputs, largely from produced waters. Transportation of petroleum, including pipeline, tank vessel and coastal facility spills, accounted for 3.5% of inputs. The category " Consumption of Petroleum" included land-based river and runoff, recreational marine vessels, operational discharges and jettisoned aircraft fuel; these accounted for 32% of total inputs. The NRC noted that urban runo ff and recreational boating require attention because the spills are chronic and often occur in sensitive ecosystems.

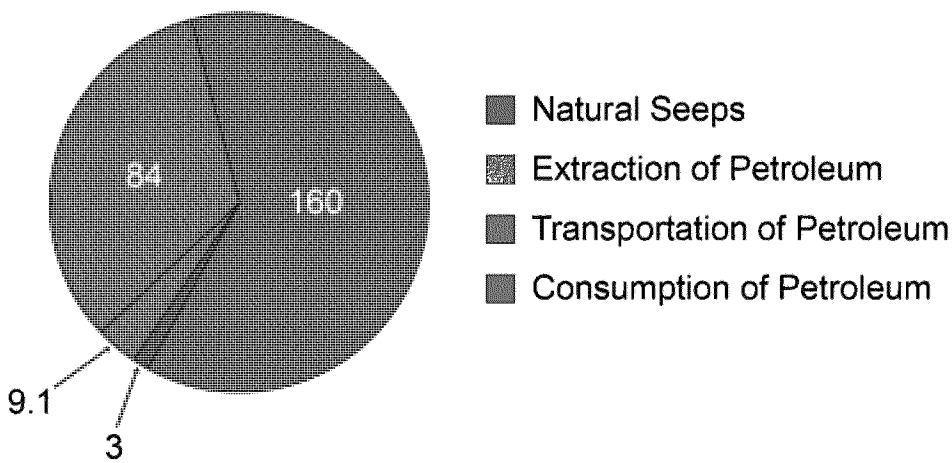

Figure 1.3 Average annual releases of oil (in kilotonnes) into North American marine waters from various sources (1990-1999). *Data from NRC (2003)*

1.4.3 Inland Freshwater Spills

Compared to marine oil spills, inland oil spills have received less attention, but freshwater spills are common, with more than 2,000 oil spills, on average, taking place each year in the inland waters of the

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continental United States (Owens et al. 1993). Although freshwater spills tend to be smaller volumes than their marine counterparts, they can have a greater potential to pose risks to the environment because of the greater likelihood that they occur within populated areas close to waterbodies with much less dilution and dispersion capacity, and involve shorelines that are often immediately adjacent to or directly impacted by the spill.

There are several recent examples of the substantial impacts of spills into inland waters. The spill of about 1,500 tonnes of Bunker C heavy fuel oil into Wabamun Lake, AB, in 2005 (due to a train derailment) resulted in residents being advised to avoid all use of lake and well water, and in impacts to habitat for numerous aquatic organisms and waterfowl (Thormann and Bayley 2008; Martin et al. 2014). The release of 26,000 bbl (1 bbl = 3,100 tonnes) of dilbit into Talmadge Creek, a tributary of the Kalamazoo River near Marshall, MI, in July 2010, resulted in the largest inland oil spill and one of the costliest spills in U.S. history. Another major, high visibility inland spill occurred in March 2013, when an Exxon-Mobil pipeline carrying Canadian Wabasca heavy crude from the Athabasca oil sands ruptured in Mayflower, AR, releasing 5,000 to 7,000 bbl (600 to 880 tonnes) of oil mixed with water into a residential neighbourhood. The oil eventually flowed through storm drains into Lake Conway, a recreational fishing lake.

Data on spills into fresh water in Canada were available primarily from the Transportation Safety Board of Canada (TSB), the NEB, the Alberta Energy Regulator (AER), and the AER's predecessor, the Energy Resources Conservation Board (ERCB). Data from the other major oil producing province, Saskatchewan, were also of interest to the Panel. However, the hundreds of spill reports in the Government of Saskatchewan Petroleum and Natural Gas Spill Report Directory (Government of Saskatchewan 2015) were not filed in a manner amenable to searching by keyword in order to focus on volume, type of spill or receiving environment. The only search criterion was spill location. Therefore, an efficient and effective review of the number and consequence of spills in Saskatchewan was not possible.

1.4.4 Spills from Pipelines

The volumes of releases of crude oil from pipeline accidents are usually small and releases are rarely into waterbodies. The TSB reported that over the 2004-2013 period, 14 pipeline accidents resulting in releases of crude oil occurred in Canada; eight of these releases were less than 1 m^3 (<1 tonne), two were between 1 and 25 m^3, three were between 26 and 1,000 m^3, and one release was over 1,000 m^3 (TSB 2013b). The AER reported that the number of crude oil pipeline incidents per 1,000 km of pipe ranged from 1.5 to 3.1 during the period 1990-2000 and from 1.0 to 1.9 during the period 2001-2012 (AER 2013). The AER stated that the stable spill frequencies in recent years may indicate that current practices may not result in further improvements; rather, new technologies or management strategies would be required to achieve significant spill reductions.

Although the incidence of crude oil releases from pipelines into fresh water have been relatively rare, spills entering rivers or wetlands have caught the public's attention due to the potential ecological damage on sensitive sites. Some of these spills are summarized in **Table 1.3** to provide context regarding volume, the type of receiving environment and environmental effects (if noted by the investigators).

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Table 1.3 Examples of recent larger-volume pipeline spills in Alberta

Date and Location	Volume and type of spilled oil	Type of Receiving Environment	Environmental Effects	Reference
January 2001, Enbridge pipeline at Hardisty, AB	3,800 m^3 crude oil	Permanent slough	3,760 m^3 recovered; no effects noted	TSB (2001)
April 2011, Plains Midstream pipeline near First Nations Community of Little Buffalo, AB	4,500 m^3 crude oil	Beaver ponds and muskeg	Beaver, amphibian, bird and small mammal mortalities; reclamation required including re-vegetation	ERCB (2013)
May 2012, Pace Oil and Gas pipeline near Rainbow Lake, AB	800 m^3 crude oil	Wetlands	No wildlife or aquatic life mortalities noted; remediation efforts reported as being effective with vegetative regrowth noted	AER (2014a)
June 2012, Plains Midstream pipeline near Sundre, AB	460 m^3 crude oil	Red Deer River	Effects on water supplies of two communities and on recreational uses; effects on wildlife, soils and riparian vegetation	AER (2014b)
July 2015, Nexen Energy pipeline at its Long Lake operations, AB	5,000 m^3 bitumen emulsion	Muskeg adjacent to the pipeline right-of-way	Duck and frog mortalities; concerns focus on effects of high-salinity process water in the spilled oil: water emulsion	AER news release https://www.aer.ca/compliance-and-enforcement/nexen-long-lake

1.4.5 Other Releases from Production Sources into Fresh Water

Crude oil releases to fresh water at production sources can be due to blowouts or "flow-to-surface" incidents associated with hydraulic fracturing ("fracking") or with *in situ* oil sands production facilities. Well blowouts can involve releases of oil, produced water, frac fluids and/or gases. A review of investigation reports by the AER and previously the ERCB showed that the main consequences were to workers and public safety due to fire, explosion and toxic gases (particularly hydrogen sulphide gas). Effects on soils and vegetation in the immediate vicinity were also noted. Only one case was found (for the period 2006-2015) where release to a waterbody was noted; in this case, wellbore fluids (produced water and diesel fuel) entered an unnamed watercourse about 100 m from the well (ERCB 2011). The volumes and/or consequence of the release to the watercourse were not discussed.

A documented release of crude oil to a surface waterbody due to a fracking incident was not found in the record of reports released by the AER or ERCB. The ERCB investigated one incident that resulted in a 'misting' of released fluids on trees and soils; however, no release into water was noted (ERCB 2012).

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The release was caused by hydraulic fracturing operations affecting a nearby producing oil well, resulting in a release of frac and formation fluids, produced water and natural gas.

The use of high-pressure steam to release bitumen from oil sands formations *in situ* can result in release of oil to the surface environment. For example, a series of four bitumen releases occurred at the Canadian Natural Resources Ltd (CNRL) Primrose high-pressure cyclic steam stimulation operations near Cold Lake, AB, from 2013-2014. The events were related to the use of high-pressure steam in a situation where vertical hydraulically-induced fractures could propagate, the cementing of wellbores was inadequate, and natural fractures and faults contributed additional pathways to the surface (Independent Panel Review 2014). Estimates of the total volume released ranged from about 1,200 to 1,900 m^3 (volume reports varied between CNRL, AER and third-party reviewers). Beaver, migratory bird, amphibian and small mammal mortalities occurred within the affected wetland complexes. A 20-ha portion of a lake had to be drained to remove bitumen and re-filled. The AER prohibited future high-pressure steaming at specific CNRL Primrose locations. CNRL reduced steam volumes and implemented enhanced monitoring and response to any further bitumen releases pending a final report from the AER.

1.4.6 Spills Due to Train Derailments

The recent increase in transport of crude oil by rail has resulted in an accompanying increase in rail-related spills. The TSB's most recent report on railway occurrences stated that over the 2004-2014 period, an increase in accidents involving release of crude oil was concurrent with an increase in shipments of crude oil by rail, from 500 car loads in 2009 to 160,000 car loads in 2013 (TSB 2015a). Five rail accidents resulted in the release of crude oil in 2013, including the Lac Mégantic derailment (see below). One accident involved a release of crude oil in 2014.

According to the U.S. Pipeline and Hazardous Materials Safety Administration (PHMSA), the United States has also experienced an increase in train-related oil spills. Between 1975 and 2009, no reported oil spills occurred during eight of those years, and spills of just one gallon or less were reported in five other years. However, more recently, data compiled by PHMSA reported a total of 3,700 tonnes of crude oil spilled from rail cars in 2013, greater than the 2,600 tonnes reported during the previous 37 years combined. A significant portion of the increased crude-by-rail transportation can be related to shale oil production in North Dakota, which has greatly expanded its output since the discovery of a new oil field in 2006 (the Parshall Oil Field producing from the Bakken and Three Forks Formations in the Williston Basin) (LeFever 2008)

Bakken crude oil produced in North Dakota was involved in three crude-by-rail spills during 2013, including the derailment in Lac-Mégantic, QC, on July 6, where a subsequent explosion and fire killed 47 people. An estimated 100 m^3 of spilled crude oil entered Mégantic Lake and the Chaudière River via surface flow, underground infiltration and sewer systems (TSB 2014b). In November 2013, derailment of 26 tank cars in Aliceville, AL, resulted in the release of nearly 2,400 tonnes of Bakken crude oil; the U.S. National Transportation Safety Board (NTSB) noted that the crude oil was released into a wetland (NTSB 2014). The third major spill occurred on December 30 near Casselton, ND, where several tank cars ruptured when a crude oil unit train derailed after striking another derailed freight train, spilling an estimated 1,300 tonnes of crude oil (NTSB 2013). A post-accident fire created dense, toxic smoke that forced a temporary evacuation of the town (NTSB 2014). The NTSB preliminary report did not note any release to surface waterbodies from the Casselton spill, and the online NTSB docket did not include any references to reports regarding effects on waterbodies (NTSB 2015).

As reported in the public media, releases of crude oil due to rail accidents continued in 2015. In several cases, the oil caught fire and towns were evacuated. Most spills occurred on land rather than in aquatic environments, but one entered a river. Examples during the first six months of 2015 include the following:

- 16 February – Canada – A Canadian National Railway (CN) freight train transporting crude oil derailed 80 km south of Timmins, ON. Of the 100 cars in the train, 29 derailed and seven crude oil tank cars caught fire. No injuries occurred and the burning oil was contained to the area (Mangione 2015).
- 16 February – U.S. – A CSX freight train hauling 109 tank cars of Bakken crude oil derailed near Mount Carbon, WV. At least seven crude oil tank cars caught fire, forcing the evacuation of about 200 people, one injury and no deaths (Van Pelt 2015). About 685 m^3 of crude oil were subsequently recovered by CSX and about 9,070 tonnes of soil were removed and shipped for disposal (Raby 2015).
- 5 March – U.S. – A BNSF oil train derailed in a rural area near Galena, IL. Of the 105 cars being transported, 21 containing Bakken formation crude oil left the track and caught fire. No injuries were reported. The U.S. EPA reported that sampling of surface water did not detect any oil in the Mississippi or Galena Rivers (EPA 2015).
- 7 March – Canada – A CN freight train derailed near Gogama, in northern Ontario. Forty of the 100 cars derailed, with five entering the Makami River and seven carrying crude oil catching fire and burning for several days (TSB 2015b). Cleanup crews deployed booms to soak up floating oil, as well as vacuum trucks to recover oil residues in the river. Thousands of tonnes of oily snow were removed. A drinking water advisory issued for Minisinakwa Lake was lifted on June 10, 2015, after analyses confirmed that hydrocarbon concentrations did not pose a risk to human health (Gillis 2015).
- 6 May – U.S. – A BNSF train derailed near Heimdal, ND, which ignited a crude oil fire in six tanker cars and forced the evacuation of approximately 40 nearby residents. No injuries or fatalities were reported (Riordan Seville et al. 2015).

1.4.7 Spills Due to Truck Transport Accidents

The total volume of crude oil transported by truck in Canada is an order of magnitude smaller than transport by pipeline, but the number of incidents per million m^3 of crude transported is four-fold greater (**Table 1.4**). The rate of incidents per volume transported was lowest for rail transportation. This disparity remained consistent in 2014, when only one crude-by-rail spill occurred while transporting a total volume of 9,339,576 m^3 of oil (significant figures are those reported by the data sources), an incident rate of 0.11 per million m^3 (data from National Energy Board [2015] and TSB [2015 a]) versus a thirteen-fold greater rate for oil transport by truck in 2011 (**Table 1.4**).

Table 1.4 Number of accidents, volume shipped and accident rate by mode of crude oil transport [a,b]. *Data from Young (2014)*

Year	Mode	Number of Accidents	Volume Shipped (m^3)	Rate (per million m^3)
2011	Truck	45	31,459,085	1.43
2011	Pipeline	259	736,285,714	0.35
2012	Rail	1	3,908,266	0.26

a, Volume data from Statistics Canada and Transport Canada; accident numbers from Transport Canada for rail and truck and from provincial governments and NEB for pipeline. Volumes were estimated using crude oil density of 0.9 kg/l. There were no in-transit rail accidents in 2011; therefore, 2012 data were used for comparison. Pipeline volume includes some diluent.
b. The number of significant figures in the table are those reported by Young (2014)

Data from the U.S. indicate a similar pattern. The volume of oil spilled per distance transported has been greatest for tanker trucks among all modes of transport since 1996 (**Figure 1.4**). However, this metric has

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decreased over the same time period for all spills from all modes of transportation, including oil spills from trucks.

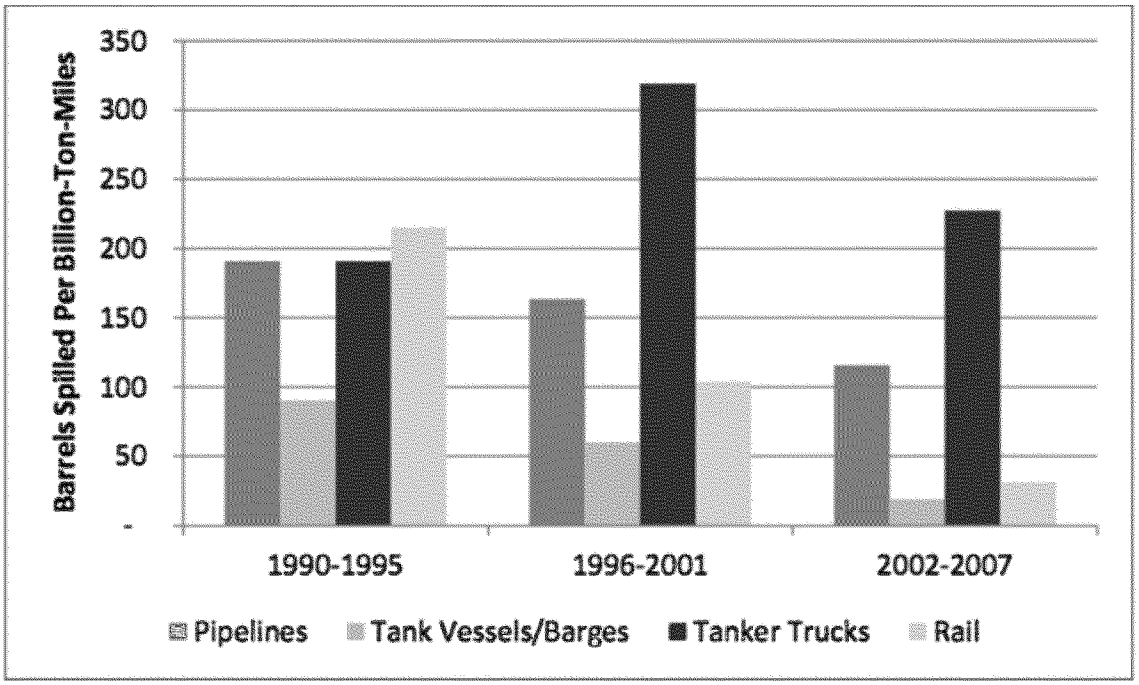

Figure 1.4 Average volumes of crude oil and petroleum product spill ed per distance transported by all modes in the United States. *Notes: Pipelines include onshore and offshore pipelines. The time periods were chosen based on the available annual data for bot h spill volume and ton -miles. The values for each time period are averages of annual data for each six-year period. Source: Frittalli et al. (2014)*

Although the incident rate is highest for truck transport, the average spill volume per incident is much smaller and mainly on land. Unlike other modes of transport, trucks are primarily used to transport oil for relatively short distances. However, a recent increase i n truck oil shipments has occurred , which is not reflected in the historic data shown in Figure 1.4. Shipment of oil by truck from shale formations in North Dakota and oil sands in Canada to U.S. refineries increased by 38% between 2011 and 2012 (Christopherson and Dave 2014). Risks to public health and safety from a tanker truck spill would be greater in the event of a fire and explosion if the spill occurred close to inhabited areas, and risks to the environment would be greater if the spill was released directly into waterbodies. For example, in 2013 a tanker truck carrying 35 m^3 of jet fuel tipped over into Lemon Creek, BC (a tributary of the Slocan River) resulting in an evacuation order affecting about 1,500 people, a "do not drink" water order that la sted a week, some fish and bird mortalities, and effects on water and sediment quality (CBC 2013; Executive Flight Centre 2014). While this incident did not involve crude oil, it illustrates the effects of hydrocarbon spills directly into waterbodies. A release of crude oil would be expected to have longer-term impacts due to the persistence of the non-volatile components of the spill in water, sediments and biota.

1.4.8 Summary of Crude Oil Spills in Canada

Spills of crude oil into marine or freshwater systems i n Canada from oil production facilities, tankers, pipelines, rail and truck transport are infrequent, and the probability of spills decreases with increasing spill size (**Table 1.5).** However, as pointed out by SL Ross Environmental Research (2014), while statistics for spills into the marine environment are readily available in Canada, statistics for inland spills into fresh water are not. The difficulty in locating inland spill data produced the "nd" results for rail and road incidents in Table 1.5. The Panel's review confirmed the difficulty of obtaining clear and reliable

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data on crude oil spills into fresh water. For example, the information presented in the sections above on spills from production facilities, rail and truck transport had to be gleaned from details within statistical reports on all incidents, whether involving oil transport or not (e.g., the TSB [2015] report on railway occurrences), by laborious searching through individual investigation reports (e.g., individual TSB or AER reports) and/or from media reports. Even when reports were available on spill incidence and volume (e.g., the AER [2013] report on pipeline performance), spills of crude oil were not always distinguished from refined product spills; instead, statistics were presented on "hydrocarbon liquids".

Table 1.5 Annual frequency of oil spills in Canada by source and volume *. Source: SL Ross Environmental Research (2014)*

Source	Spill Volume (m^3) <1	1-10	10-100	100-1,000	1,000-10,000	> 10,000
Offshore exploration and production[a]	20.9	1.7	0.4	0	0	0
Vessel Spills[b]						
Crude	nd[d]	nd	0.02	0.01	0.19	0.004
Refined	nd	nd	0.60	0.10	0.024	0.00
Fuel	nd	nd	1.90	0.60	0.01	0.00
Total	nd	nd	2.52	0.71	0.05	0.004
Pipelines[c]						
Release into	0.56	0	0.20	0	0	0
Total	0.75	3.8	1.89	0.94	0	0
Rail	nd	nd	nd	nd	nd	nd
Road	nd	nd	nd	nd	nd	nd

[a]. *2004 to 2013. Canada-Nova Scotia Offshore Petroleum Board (http://www.cnsopb.ns.ca/environment/incident-reporting) and http://www.cnsopb.ns.ca/environment/incident-reporting*
[b]. *WSP Canada Inc. (2014).*
[c]. *2008 to April 2013. https://www.neb-one.gc.ca/sftnvrnmnt/sft/dshbrd/index-eng.html*
[d]. *nd, no data; information was not publically available or was not immediately accessible.*

Notwithstanding the apparent relative infrequency of crude oil spills into aqueous environments in Canada, the consequences of spills into sensitive waterbodies can be substantial, both with respect to impacts on human health, safety and the environment and with respect to economic impacts. Therefore, a primary focus of this Report is on the consequences of spills, no matter how low the probability may be. This focus is commensurate with the key questions posed by the sponsors and stakeholders.

1.5 Organization of This Report

The relationships among the chapters of this report and the key questions posed by the sponsors and stakeholders are illustrated in **Table 1.6**. There is overlap in the subject matter among the chapters but the Panel strove to avoid duplication. In some cases, topics are discussed from a different perspective, or in more or less detail, depending upon the primary focus of the chapter. Chapter 8 considers all of the major

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topics within a risk framework and uses case studies to examine the importance of physical, chemical and biological factors, as well as cleanup and recovery measures in determining short-term and long-term risk of spills. The final chapter (Chapter 9) presents the Panel's recommendations.

Table 1.6 Relationship between chapters and key questions

Chapter	**Question 1:** Behaviour of Crude Oil Types	**Question 2:** Chemical Composition and Toxicity of Crude Oil Types	**Question 3:** Microbial Processes Affecting Crude Oil Toxicity and Characteristics	**Question 4:** Relationship Among Crude Oil Behaviour, Toxicity, Characteristics and Remediation	**Question 5:** Research Priorities	**Question 6:** Spill Prevention, Preparedness and Response
2: Composition, Properties and Behaviour	X	X	X		X	
3: Effects of Environment on Oil Fate and Behaviour			X	X	X	
4: Toxicity and Ecological Relevance		X	X	X	X	X
5: Modeling Spill Behaviour				X	X	
6: Spill Response Options					X	X
7: Prevention and Response Decision Making					X	X
8: Risks from Oil Spills	X	X	X	X	X	X

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CHAPTER 2: CHEMICAL COMPOSITION, PROPERTIES AND BEHAVIOUR OF SPILLED OILS

Abstract

The chemical composition of petroleum [1] is of utmost importance for understanding oil spills in aquatic environments because the chemistry of the oil dictates its physical properties (e.g., density and viscosity), behaviour (e.g., spreading, sinking, dispersion), biological impacts (e.g., toxicity, susceptibility to biodegradation) and ultimate fate in the environment. Thus, petroleum chemistry is important to oil spill responders for predicting hazards, such as fire and explosion, and for selecting suitable cleanup and remediation approaches. It is also important to regulators for forensic purposes, monitoring and determining acceptable endpoints for spill remediation.

The petroleum hydrocarbons transported within Canada range from ultra-light condensates and light oils to heavy oils and bitumens, as well as blends like diluted bitumens and heavy fuel oils. Each oil is a complex mixture of thousands of chemicals, not all of which are hydrocarbons, and every oil has a unique chemical 'fingerprint'. Conventionally, petroleum is separated into four major fractions for analysis: saturates, aromatics, resins and asphaltenes. Whereas the structures of many saturated and aromatic petroleum compounds are known and have been studied individually or within an oil matrix, the chemically diverse, high molecular weight resins and asphaltene fractions have resisted characterization. Additional minor constituents in oil may include sulphur (e.g., as hydrogen sulphide gas), naphthenic acids, metals and minerals.

Spilling oil onto water progressively changes its chemical fingerprint through physical, chemical and biological processes collectively called 'weathering'. As oil spreads on the water surface, small molecules evaporate, others may be oxidized by sunlight, and the oil may form emulsions with water. In the water column oils may disperse as droplets and/or sink and/or form 'tar balls' after interacting with suspended particles; light components may dissolve in the water; others may be selectively biodegraded by microbes. Oil that reaches the shoreline or sediments may become sequestered and/or re-emerge over time, but oil interactions with ice are currently not well known. Measurable changes in the 'fingerprint' of spilled oil over time reflect its initial chemistry and the magnitude of weathering processes and therefore can be used to monitor remediation progress and determine endpoints. Importantly, just as each crude oil is chemically unique, each oil spill reflects the specific environmental conditions at the spill site acting on a particular oil over a given time period. The diverse petroleum types form a continuum of chemical compositions, physical properties and spill behaviours.

To provide background for later chapters on oil fate (Chapter 3), toxicity (Chapter 4), modeling (Chapter 5) and remediation of oil spills (Chapter 6), as well as spill response decision-making (Chapter 7), this chapter summarizes the common chemical components of crude oils, describes a suite of crudes representing the compositional range of oils commonly transported in Canada, and briefly outlines the general behaviour of oil spilled in marine and freshwater environments. Further research is needed to fully characterize the properties and spill behaviours of emerging oil types, including diluted bitumen blends and unconventional oils such as shale oil.

[1] *The terms 'petroleum' and 'crude oil' are considered here to be synonyms and are used interchangeably.*

2.1 Chemical Composition of Oils

Crude oils are naturally -occurring complex mixtures of thousands of individual compounds, but primarily they comprise hydrocarbons (molecules consisting of only carbon and hydrogen) and lesser proportions of compounds containing heteroatoms (e.g., nitrogen, sulphur and/or oxygen) in addition to carbon and hydrogen. Commonly, small amounts of metals , minerals and sometimes inorganic sulphur are also present. The majority of petroleum components are derived from organic matter from ancient aquatic plants, animals and microbes, and their proportions in the oil reflect the source(s) and geological history of that oil during its formation.

> Crude oils are highly complex, diverse mixtures of chemicals. Each oil has a unique composition or 'chemical fingerprint' that usually consists of four major classes of chemicals: Saturates, Aromatics, Resins and Asphaltenes (SARA) and sometimes minor constituents like metals, sulphur, solids and water.

2.1.1 Four Major Chemical Fractions of Crude Oil

Most crude oil constituents can be divided into four major chemical classes known by the acronym SARA: Saturates, Aromatics, Resins and Asphaltenes, as described below and illustrated in **Table 2.1**. Historically, these groups are defined as solubility fractions by liquid-solid chromatography, as explained in Appendix A (**Figure A1**). Appendix A also presents an overview of analytical methods commonly used to chemically characterize oils . Together thes e fractions contribute to the bulk properties of petroleum, such as viscosity, specific gravity and susceptibility to biodegradation, as discussed in detail below (Section 2.2).

It is important to note that **Table 2.1** presents only a few examples of the thousands of possible structures of petroleum components. Indeed, the presence and positions of one or more alkyl groups (e.g., $-CH_3$, $-C_2H_5$) on parent structures, whether saturated or aromatic, can produce myriad 'isomers' (chemicals having the same chemical formula but different structures) and 'homolog ous series' of similar chemicals that differ by the number of alkyl substituents. Isomers and members of homologous series may have very different adsorption properties, water solubility and susceptibility to biodegradation. A paraffin with four carbon atoms can exist in only two isomer forms, but one with 14 carbon atoms theoretically can exist as 60,523 isomers (Speight 2014). Multiple isomers cause technical problems for peak resolution by gas chromatography (GC) and contribute to the unresolved complex mixture (UCM) or 'hump' in gas chromatograms (Appendix A).

2.1.1.1 Saturates

This chemical class is typically the major component of petroleum. It exclusively comprises hydrocarbon molecules having single carbon - carbon bonds, with all remaining bonds being saturated with hydrogen atoms (i.e., no double - or triple-bonded carbon). Three subclasses are defined within this fraction: *paraffins* and *isoparaffins* having straight or branched chain structures, respectively; *naphthenes* (*cycloparaffins,* also called *alicyclic hydrocarbons*) having one or more saturated ring structures; and fused-ring aliphatics, including *hopanoids* and *steroids,* having large, complex structures combining paraffinic and naphthenic structures (**Table 2.1**). Olefins (partially unsaturated hydrocarbons), such as alkenes and alkynes, are scarce or absent in petroleum (Speight 2014).

> The saturate fraction is considered the least toxic of the four major petroleum fractions and the most readily biodegradable, with the exception of steroids and hopanoids that tend to resist biodegradation.

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Petroleum paraffins range from gases and vapours of one to five carbon atoms (C_1-C_5) under normal pressure (e.g., methane gas, CH_4) to liquids of C_5-C_{16}, to crystalline waxes of $C_{>20}$. Normal (*n*-)alkanes have straight chains with no branching (i.e., no alkyl substitutions on the hydrocarbon backbone) and therefore have the general formula C_nH_{2n+2}. *iso*-Alkanes are branched structures with alkyl substitutions at single or multiple points on the backbone, creating numerous series of isomers having the same molecular weight but different structures.

In general, *n*-alkanes $<C_5$ are highly volatile (and therefore not likely to remain in the water column for biodegradation, except at high pressures in deep water), and those between C_5 and $\sim C_{30}$ are biodegradable when present in crude oil. Therefore, *n*-alkanes of $\leq C_{30}$ are unlikely to persist in the environment and pose little toxicological risk to aquatic life, unless they are ingested in oil droplets, because they are essentially water-insoluble. *iso*-Alkanes tend to be more persistent in the environment because they are more slowly degraded than their *n*-alkane counterparts; in fact, pristane (**Table 2.1**) and phytane, which are common and often prominent *iso*-alkanes in petroleum (Appendix A, **Figure A2**), have been used as short-term petroleum biomarkers because they usually resist biodegradation until the *n*-alkanes have been depleted (Head et al. 2006). Therefore, petroleum biomarkers can be used as internal standards to monitor the biodegradation rates of susceptible oil components, or for forensic evidence to identify sources of contamination in the environment. For example, triaromatic steroids containing aromatic and aliphatic moieties (**Table 2.1**) were used to distinguish naturally-occurring background hydrocarbons from diluted bitumen (dilbit) contamination in the Kalamazoo River spill (US-EPA 2013a; Chapter 8). See Chapter 7 for a more complete discussion of petroleum biomarkers.

> The term 'biomarker', as used in petroleum chemistry, designates a chemical compound originally derived from living organisms that is poorly biodegradable and therefore persists over geological time in the oil. This differs from the definition of 'biomarker' used in environmental toxicology (Chapter 4). Commonly used petroleum biomarkers include certain iso-alkanes and complex cyclic alkanes.

Naphthenes, or *cyclo*-alkanes, have the general formula C_nH_{2n} and can exist in multiple fused ring structures with single or multiple alkyl substitutions. Low molecular weight (LMW) *cyclo*-alkanes ($\leq C_6$) are volatile and can cause toxic effects (e.g., US-EPA 1994; Sikkema et al. 1995) because they can partition into biological membranes. As with *iso*-alkanes, *cyclo*-alkanes tend to be more resistant to biodegradation than *n*-alkanes of the same molecular weight (Head et al. 2006). Diamondoids (based on the adamantane structure; **Table 2.1**) are examples of multicyclic aliphatics, having cage-like structures with or without alkyl side-groups (Wei et al. 2007). They are found in significant concentrations in bitumen and are less susceptible to biodegradation than their polycyclic aromatic hydrocarbon (PAH) counterparts (Government of Canada 2013).

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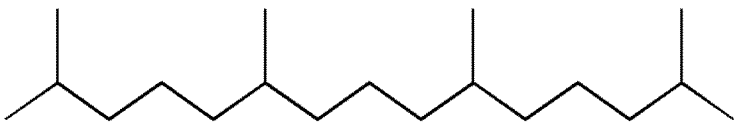
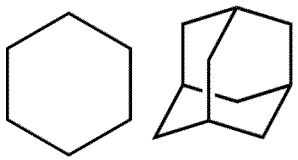
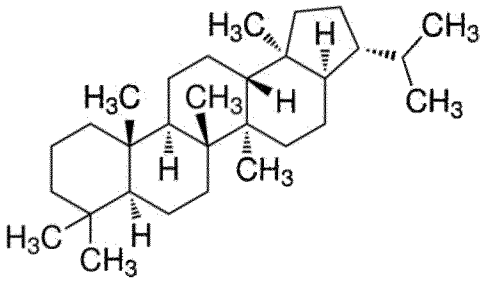
Table 2.1 Major chemical classes comprising petroleum, and structures of some example molecules

Name of chemical class or fraction	Example of class or fraction	Structure of example compound(s)	Properties and relevance of class or fraction
SATURATES			Usually the most abundant chemical class in petroleum; highly water-insoluble; typically are biodegradable and generally non-toxic
n-Alkanes	*n*- Hexadecane		*n*-Alkanes <C_6 are light liquids or gases and may be toxic; C_6-C_{18} are readily biodegradable; >C_{20} are waxes and more difficult to biodegrade
iso-Alkanes	Pristane		*iso*-Alkanes are more resistant to biodegradation than *n*-alkanes; some, such as pristane and phytane, are used as short-term biomarkers for aerobic biodegradation
cyclo-Alkanes and Diamondoids	Cyclohexane; Adamantane		More resistant to biodegradation than *n*-alkanes; diamondoids may be used as robust biomarkers during biodegradation (de Araujo et al. 2012)
Hopanoids and Steroids	17α(H),21β(H)-Hopane (30αβ)		Many are non-biodegradable and therefore are used as long-term biomarkers for forensics and biodegradation assessment

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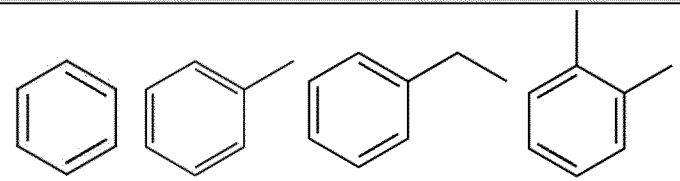
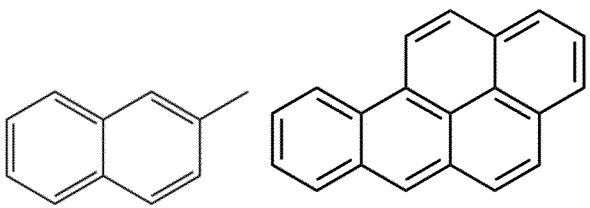
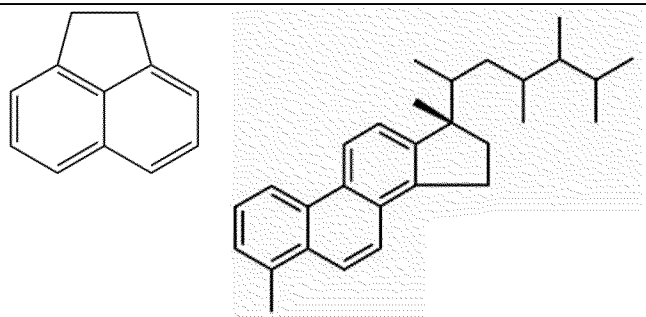
Name of chemical class or fraction	Example of class or fraction	Structure of example compound(s)	Properties and relevance of class or fraction
AROMATICS		The examples shown below are strictly hydrocarbons; however, some aromatics having heteroatoms (i.e., S-, N- and O-heterocycles) fractionate with the aromatic hydrocarbons and are often considered together functionally. Examples of such aromatic heterocycles are shown in the resins fraction below.	A hydrocarbon class that is generally more toxic than saturates; many isomers and homologous series are possible when aliphatic groups are attached to the aromatic rings.
Monoaromatics	BTEX: Benzene, Toluene, Ethylbenzene and Xylene isomers (*ortho*-xylene shown)		The BTEX series is volatile, more water-soluble than other hydrocarbons, acutely toxic and/or carcinogenic, and relatively biodegradable under aerobic and anaerobic conditions
Polycyclic aromatic hydrocarbons (PAHs) and alkylated series (alkyl PAH)	2-Methylnaphthalene and Benzo[a]pyrene		Large number of isomers are possible due to multiple positions for alkyl side chains; some PAHs are toxic and/or carcinogenic; many persist in the environment because they resist biodegradation; benzo[a]pyrene is a US-EPA 'Priority Pollutant'
Naphtheno-aromatics; aromatic steroids	Acenaphthene and a triaromatic steroid		Combination of saturated and unsaturated cyclic hydrocarbons; acenaphthene is a US-EPA 'Priority Pollutant'; triaromatic steroids have been used for forensic attribution of environmental contamination

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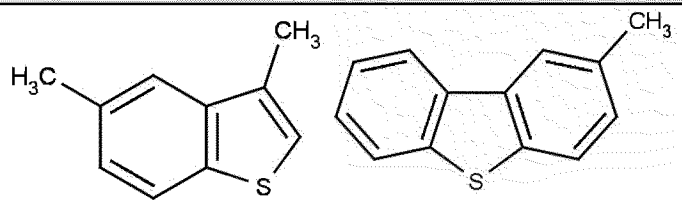
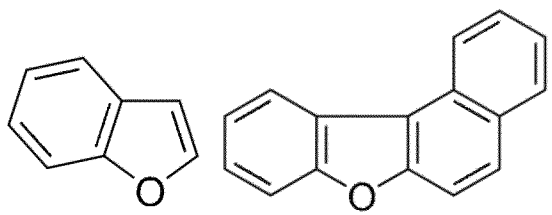
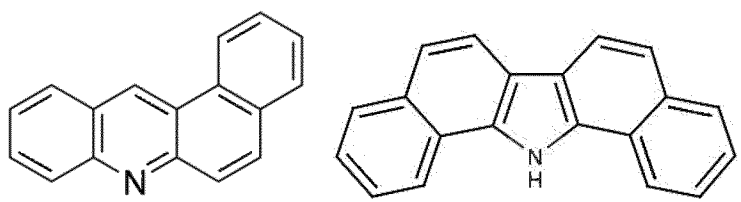
Name of chemical class or fraction	Example of class or fraction	Structure of example compound(s)	Properties and relevance of class or fraction
RESINS		The structures of individual resin compounds have not been determined, but they contain one or more S-, N- and/or O-heteroatoms; examples of aromatic S-, N- and O-moieties that may form part of resin structures are shown below. The individual chemicals below are commonly considered to be part of the Aromatics fraction (despite being heterocycles rather than hydrocarbons).	A solubility class related to asphaltenes (below), but of lower molecular weight, lower aromatic content and greater polarity; poorly detected and resolved by most GC-based methods
Examples of S-containing moieties that may be constituents of resin molecules	Moieties may include: Dimethylbenzothiophene and 2-Methyldibenzo-thiophene		Organic sulphur moieties such as these may contribute to total sulphur content of sour crude oils
Examples of O-containing moieties that may be constituents of resin molecules	Moieties may include:Benzofuran; Benzo[b]naphthofuran		Organic oxygen-containing groups in resins may be generated from PAHs by photooxidation or partial biodegradation
Examples of N-containing moieties that may be constituents of resin molecules	Moieties may include: Benz[a]acridine;Dibenzo[a,i]carbazole		N-containing groups contribute to polarity of the resins fraction
ASPHALTENES		The structures of individual asphaltene molecules have not been determined; they are related to resins (above) but have higher molecular weight, greater aromatic content and are less polar than resins.	A solubility class of complex high molecular weight (HMW), polar compounds that are water-insoluble; contribute to heavy oil viscosity; non-biodegradable

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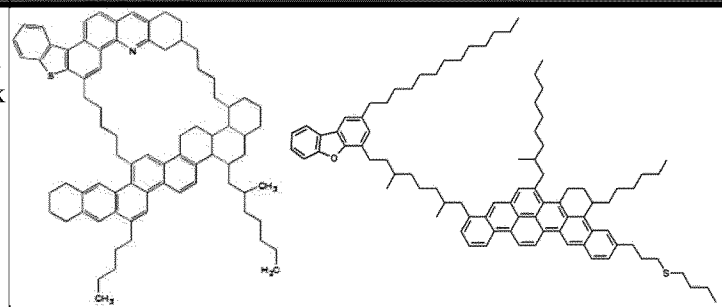
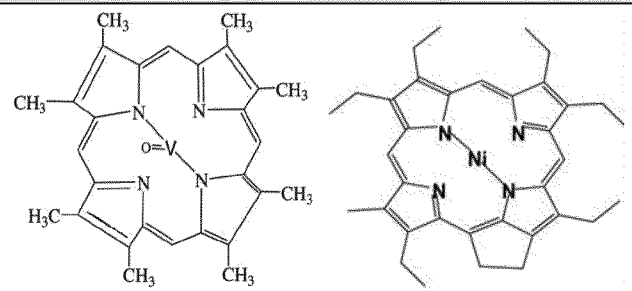
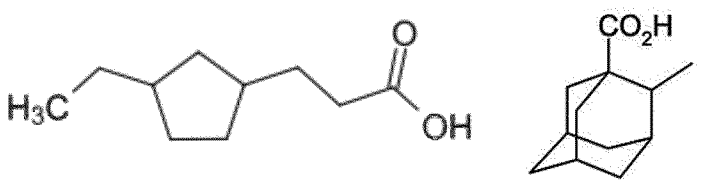
Name of chemical class or fraction	Example of class or fraction	Structure of example compound(s)	Properties and relevance of class or fraction
	Hypothetical asphaltene molecules, after Hoff and Dettman (2012) and Boek et al. (2010)		Numerous models of asphaltene structures have been proposed of different size, aromaticity and degree of condensation; different subclasses of asphaltenes with different chemical properties have been proposed recently
MINOR ORGANIC COMPONENTS			
Petroporphyrins (organometals)	C28 Etio vanadium porphyrin; Nickel deoxophyllo-erythroetioporphyrin (D PEP)		Vanadium and nickel are common metals coordinated in centre of the porphyrin structure; the core derives from (bacterio)chlorophyll molecules transformed over geological time by pressure and heat
Naphthenic acids	Ethylcyclopentane-3-propanoic acid; Diamondoid acid		Heterogeneous class of O-containing polar compounds associated with acute toxicity to aquatic life; may be products of partial biodegradation of hydrocarbons; many resist biodegradation

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Fused complex alicyclic hydrocarbons include hopanoids and steroids that are derived from microbial, animal and plant sterols and terpenes. Some of these compounds are extremely resistant to biodegradation and therefore are ideal oil biomarkers that persist for thousands or even millions of years (Peters et al. 2005). Others, such as cholestane and possibly hopane, are less stable and may slowly biodegrade (Prince and Walters 2007).

Waxes (linear alkanes >C_{20}), which can represent a substantial proportion of certain crudes, may precipitate out of petroleum at low temperature and can also form coatings on surfaces, affecting the interfacial properties of the oil (Hollebone 2015). They are non-toxic but resist biodegradation due to low bioavailability in water.

2.1.1.2 Aromatics

Aromatic hydrocarbons are cyclic, planar, unsaturated compounds having structures based on a single benzene ring or multiple fused benzene rings. They can have one or more alkyl groups in various isomeric arrangements. The most common (alkyl)-monoaromatics in petroleum are the BTEX series (benzene, toluene, ethylbenzene and the three xylene isomers; **Table 2.1**). Benzene, ethylbenzene and toluene are on the US-EPA list of toxic 'Priority Pollutants' (*http://water.epa.gov/scitech/methods/cwa/pollutants.cfm*). They are also light enough to be volatile, presenting flammability and breathing hazards.

> Monoaromatics are the most water-soluble of the hydrocarbons and thus the most mobile in the water phase by diffusion. They may cause acute toxicity because of their ability to partition into biological membranes. PAHs having two or more fused aromatic rings are less volatile and commonly more persistent than monoaromatics.

PAHs have two or more fused aromatic rings and may additionally have alkyl side-groups (alkyl PAH). Many of the smaller PAHs are of intermediate biodegradability that is inversely proportional to the number of fused rings in the structure. Several 'parent' PAHs (i.e., unsubstituted PAHs such as benzo[a]pyrene; **Table 2.1**) are on the US-EPA list of 'priority pollutants' because of their toxicity, carcinogenicity (typically via inhalation) and/or resistance to biodegradation (van Hamme et al. 2003), even though these parent PAHs typically represent <15% of the total PAH (TPAH) in the oil. The bulk of PAHs have one or more alkyl substituents in various positions (e.g., 2-methylnaphthalene; **Table 2.1**), generating many homologous series of alkyl PAH with different physical and chemical properties and sometimes very different biological effects and biodegradability (Wang et al. 1998). The chronic toxicity of oil is attributed primarily to alkyl PAHs having 3–5 rings (discussed in detail in Chapter 4).

Naphthenoaromatic structures are combinations of saturated and unsaturated rings, with or without alkyl groups. Their concentrations, along with the naphthenes, tend to be greater in heavier oils (Speight 2014) and contribute to the overall resistance of these oils to biodegradation. An example is acenaphthene (**Table 2.1**), which is also a US-EPA 'priority pollutant'.

Some heterocycles (e.g., those shown in **Table 2.1** as moieties potentially found in resin structures), including dibenzothiophenes, furans, carbazoles and their alkyl-substituted homologues, fractionate with the aromatics even though they are not hydrocarbons. Some of these aromatic heterocycles are biodegradable, whereas some isomers resist enzymatic attack (Kropp et al. 1996) or are only partially oxidized, yielding dead-end products that are more water-soluble than the parent compound and have different toxicity.

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2.1.1.3 Resins

> Resins are not hydrocarbons since their chemical structures include elements other than carbon and hydrogen, i.e., the heteroatoms S, N and/or O.

Unlike saturates and aromatics that can be defined by their chemical structures, petroleum resins (and asphaltenes, Section 2.1.1.4) are defined on the basis of their solubility in hydrocarbon solvents rather than on discrete structures (Appendix A, **Figure A1**), with the resins fraction being soluble in *n*-pentane or *n*-heptane but insoluble in liquid propane (Speight 2004). The structures of individual petroleum resins have not been determined, although they all have one or more heteroatoms that may be incorporated into ring structures that are saturated, partially saturated or aromatic, and/or long-chain paraffin residues, with or without heteroatoms. Resins are chemically related to the asphaltenes (Section 2.1.1.4), but are smaller (molecular weights estimated at 500 – 1,000 daltons [Da]; Bertoncini 2013), have lower aromaticity and are more polar[2] than asphaltenes. In the petroleum industry, resins are considered the smallest of the polar compounds in oil (versus hydrocarbons that are non-polar). Light paraffinic crude oils may comprise >97% hydrocarbons and <3% resins, whereas heavy oils and bitumen may be only ~50% hydrocarbons with the remainder being resins, asphaltenes, naphthenic acids and metals (described below). In general, the resins constitute a greater proportion of petroleum mass than the other non-hydrocarbon components, although this is not apparent by GC methods because the resins either tend to be poorly resolved, contributing to the UCM 'hump' in chromatograms (Appendix A), or not to be resolved at all by GC.

The polarity of resins is contributed by the presence of heteroatoms in diverse moieties within the resin structure, such as those shown in **Table 2.1**, i.e., S-containing aromatic thiophenes, benzothiophenes, dibenzothiophenes, naphthobenzothiophenes and aliphatic alkyl sulphides; N-containing pyrroles, indoles and carbazoles; and/or O-containing hydroxyl, ester, carboxylic acid, carbonyl, furan and sulfoxide/sulfone functional groups. These heteroatoms make resins more polar than hydrocarbons (Speight 2004) and in some cases more toxic because of their water solubility (Melbye et al. 2009). Because of the dominant effect of these moieties on the resins fraction, the resins are also sometimes called the 'Polar' or 'NSO' fraction of petroleum.

Resins typically resist biodegradation, although some may be partially oxidized by microbes to generate products of greater polarity and water solubility and thus potentially greater mobility, toxicity and environmental persistence. In fact, some resins may represent the dead-end products of incomplete biodegradation of organic matter over geological time, which are resistant to further enzymatic attack, or that may have been generated from hydrocarbons in spilled oil by poorly-understood photooxidative processes (Section 2.4.1.3).

2.1.1.4 Asphaltenes

> Asphaltenes are the most complex, most diverse and highest molecular weight components of petroleum. They are also the least susceptible to biodegradation. Asphaltenes comprise a small proportion of light crude oils and greater proportions of heavier oils and bitumens, conferring increased viscosity and density, among other properties.

Asphaltenes are not hydrocarbons because they contain S-, N- and O-heteroatoms, like the resins, but they have greater average molecular weight (>400 Da, mean weight ~1,200 Da [Gray 2015]). This chemical class (Speight 2004) represents myriad individual chemicals and is defined on the basis of insolubility in *n*-alkanes, such as pentane and heptane, but solubility in toluene (Appendix A, **Figure A1**). Thus, petroleum can be separated into

[2] *Polarity refers to the distribution of electric charge across a molecule. Simplistically, polar compounds dissolve better in water (a polar solvent) than do non-polar compounds.*

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two major fractions simply by diluting it in 40 volumes of *n*-pentane (Speight 2004) or similar solvent, whereupon the asphaltenes precipitate and the nC_5-soluble fraction (known as the maltenes or de-asphalted oil) remains dissolved. The proportion of asphaltenes in a crude oil affects its properties by causing it to adhere to and affect the wetting characteristics of surfaces it attaches to, including pores and channels in sediments, earning this class the nickname 'the cholesterol of crude oil' (Boek et al. 2010).

Despite considerable research (e.g., Strausz et al. 1992; Peng et al. 1997; Sheremata et al. 2004; Dettman et al. 2005; Hoff and Dettman 2012), the structure of individual molecules within the asphaltene class has remained controversial due to their HMW, complexity, heterogeneity and propensity to intimately associate with other petroleum components like resins (Speight 2004). Asphaltene molecules are proposed to exist as nanoaggregates (Boek et al. 2010) and/or as colloidal micelles in the bulk petroleum, with resins forming the outer layer and preventing asphaltene aggregation (Petrova et al. 2011). Such properties and interactions make chemical characterization technically difficult. Asphaltenes are neither resolved nor detected by conventional GC but instead are subjected to alternative analyses, such as gel permeation chromatography, nuclear magnetic resonance and high-temperature simulated distillation (Appendix A). Speight (2004) considered that the predominant chemical groups in asphaltenes are aromatic (sometimes >50% of the total carbon) with saturated hydrocarbon linkages. Hoff and Dettman (2012) have isolated and defined different subclasses of asphaltenes having varying degrees of condensation and differing alkyl substitutions (**Table 2.1**) and therefore different properties. Yang et al. (2015) have further inferred and differentiated structural features of asphaltenes that stabilize oil:water emulsions through interfacial activity (Yang et al. 2014). Thus, the structures and properties of asphaltenes are gradually emerging with advances in analytical techniques but consensus has yet to be achieved.

<u>Recommendation:</u> Continued fundamental research is needed into the composition of the high molecular weight, polar components of oil, and standardized analytical methods for characterizing these fractions should be developed and published. Characterization of the resins and asphaltenes classes is presently incomplete because of current analytical technical limitations preventing comprehensive, detailed monitoring and prediction of the behaviour and fate of many spilled oils, particularly heavy oils and bitumen blends that are rich in these fractions.

Because of their size and complexity, asphaltene molecules are extremely resistant to biodegradation (despite a small number of questionable literature reports purporting to demonstrate asphaltene degradation). Their stability in the environment is shown by their preservation for thousands of years in ancient artifacts and their modern use in asphalt roads and roof shingles.

2.1.2 Minor Components of Crude Oils

> In addition to SARA fractions, petroleum may contain small and variable concentrations of non-hydrocarbon components derived from biological or geological sources, such as metals, organometals, sulphur, naphthenic acids, mineral particles and /or water.

Some of the minor constituents of crude oils include organometallic compounds, such as petroporphyrins (Caumette et al. 2009) containing vanadium (V), nickel (Ni), iron (Fe) or copper (Cu) derived from plant and microbial sources and dissolved in the bulk oil, as well as inorganic salts of these metals and others in colloidal suspension (Speight 2014). Metalloporphyrin structures (**Table 2.1**) are often associated with the asphaltenes; therefore, metals are more abundant in heavier petroleum fractions and in heavy bituminous crudes at concentrations up to hundreds of parts per million (Fingas 2015b).

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Whether metal atoms that are strongly associated with their organic ligands in petroleum actually become water-soluble and bioavailable after oil spillage into water is a matter of debate (Joung and Shiller 2013). Thus, reports of environmental impacts of heavy met als from oil contamination vary. For example, concentrations of Ni and V in Saudi Arabian coastal sediments contaminated by massive volumes of Kuwait oil during the 1991 Gulf War were only slightly elevated above background values, and may have resulted from oil combustion as well as spillage (Fowler et al. 1993). In contrast, tonnes of Cu, Ni and V were estimated to have been released to the Atlantic Ocean during fuel oil leakage from the sinking of the *Prestige* oil tanker (Santos-Echeandía et al. 2008). Seawater concentrations of Cu and Ni (but not V) were elevated in the upper water column (0 -50 m depth), associated with upward movement of escaping fuel, and in deep waters that had longer contact time with the oil (Santos -Echeandía et al. 2008). Similarly , Wise Jr. et al. (2014) detected increased concentrations of chromium (Cr) and Ni in floating oil and tar balls associated with the Deepwater Horizon (DWH) blowout in the Gulf of Mexico and elevated concentrations in sperm whale tissue samples collected i n the Gulf versus control samples. In contrast, Joung and Schiller (2013) found that samples of surface waters near the DWH blowout site did not show unusual concentrations of Ni or several other metals, but deep water samples collected near the well had higher concentrations of cobalt (Co). Elevated barium (Ba) concentrations detected in deep water were likely associated with the drilling mud used to contain the blowout rather than the oil plume itself. Thus, the magnitude and effects of metal contamination from individual spills require additional study, whereas the accumulation of metals in chronically hydrocarbon -contaminated water and sediments (e.g., those impacted by refineries or shore-to-ship oil transfer) is well-known (e.g., González-Macías et al. 2006).

Inorganic S present in elemental form (S^0) or as gaseous H_2S may be a minor component by mass (often <1%), but H_2S is toxic to humans even at parts per million levels. H_2S is also a major concern regarding corrosion to infrastructure (e.g., pipelines and surface handling facilities) and explosion potential in confined spaces, including rail cars. Depending on the water content of the oil and its pH, H_2S may exist as a gas or may be dissolved in the water phase. H_2S may represent only a fracti on of the total S in sour crudes, which can contain significant quantities of organic S compounds (**Table 2.1**) that contribute to total S.The organic S compounds have different chemical properties than H_2S, and are more problematic in refineries than in the environment. Oils with a significant proportion of total S (usually ≥1%) are called 'sour' crudes, as opposed to 'sweet' crudes with ≤0.5% S (Section 2.3.6). Sweet oils are generally more valuable than sour, as the presence of S imposes a price penalty at the refinery, and H_2S concentrations must be reduced before transportation for safety reasons.

Naphthenic acids (NAs; **Table 2.1**) comprise a large, diverse group of alkyl -substituted cyclic and noncyclic carboxylic acids, historically defined as having th e chemical formula $C_nH_{2n+Z}O_2$, where *z* reflects the number of rings in the structure (Clemente and Fedorak 2005; Johnson et al. 2011). Thus, they share similarities with the resins fraction, being composed of both hydrocarbon and heteroatom moieties and being polar. NAs have also been defined by the extraction methods used to recover them from samples and by their infrared (IR) absorbance spectra, but the range of possible structures has been controversial, currently precluding a definition accepted by both industry and academia (Grewer et al. 2010; Rowland et al. 2011). Whereas NAs were previously considered to be saturated compounds with a single acidic group, this class has e xpanded to include aromatic NAs (Jones et al. 2012) and NAs with multiple oxidized functional groups (i.e., carboxylic acids plus hydroxyl groups). Definition of NAs has become even more nebulous (Grewer et al. 2010) with the development of sophisticated a nalytical methods (such as ESI FT-ICR-MS and multi-dimensional GC-MS; Appendix A) that detect more isomers and analogues than previously recognized. NAs have been used industrially as components of wood preservatives and mixtures are commercially available . However, application of new analytical methods has shown that the component NAs in those mixtures differ significantly from naturally occurring suites of NAs in petroleum -impacted waters (Grewer et al. 2010), throwing into question some of the early research on NA biodegradation conducted using commercial mixtures.

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NAs are ubiquitous, but have been associated particularly with biodegraded oil reservoir fluids and oil sands process waters in western Canada where they may represent the accumulation of dead-end products of incomplete microbially-mediated oxidation of *cyclo*-alkanes (naphthenes; **Table 2.1**) (Rowland et al. 2011) over geological time. This view is supported in part by NA persistence in the environment, as many resist further biodegradation (Whitby 2010). Current environmental interest in NAs arises from their reported acute toxicity to aquatic life (Allen 2008; Zhang et al. 2011), contribution to Total Acid Number (TAN; the petroleum acidity parameter), and accretion of precipitates (scale) that increase corrosion of pipelines, refinery units and other oil-handling infrastructure. Due to their acidic group(s), they are more polar than hydrocarbons, and this water solubility makes them more mobile in the environment than the oil itself. In fact, petroleum that has been 'water-washed' in the reservoir or the environment may become depleted in NAs as they dissolve into the water fraction. Conversely, spilled oil having high NA contents may be a source of toxicity in aquatic environments as the NAs dissolve in water. However, attribution of toxicity to specific NA compounds within a complex mixture of NAs is currently in flux due to the expanding definitions of NA structures, discussed above. Further discussion of NA toxicity in oil sands-associated environments is available (Gosselin et al. 2010).

Crude oils may also contain small amounts of minerals, such as clays, and emulsified water originating either from the geological formation (connate water) or from water injected into the reservoir during oil recovery (produced water). The great majority of entrained or emulsified water is removed from the oil to meet pipeline specifications (typically ≤0.5% water plus sediments).

It is clear that the chemical composition of crude oils is extremely complex and variable, and the structures of many oil components, including the UCM, are currently cryptic. The limitations of resolving and identifying individual molecules in the resins and asphaltenes fractions, particularly, are slowly being addressed with renewed scientific interest and development of new techniques. However, improved analytical methods for chemical characterization are still needed to gain a better understanding of the persistence and behaviour of these components in spilled oil. Additionally, the quality of data in different oil property archives varies according to methodology; development and adoption of standardized methodology with data quality indices would allow for comparisons among studies.

<u>Recommendation:</u> Additional analytical research is needed over the long-term to address gaps in fundamental knowledge of asphaltenes, resins and naphthenic acids, and to populate oil-property databases with verified data. Such knowledge provides insight into criteria that affect emulsification, photooxidation, biodegradation, toxicity, adhesion, etc., particularly for heavy oils and bitumens that have high proportions of such chemical classes.

2.2 Bulk Properties of Oil

Complex mixtures, such as petroleum, exhibit complex chemical and physical properties requiring suites of analytical methods to discern important differences between oils. Several properties are discussed below; they are interrelated because all are affected by oil composition. Appendix B (**Table B1**) comprises a summary of major oil properties, associated analytical methods and the relevance of those properties to oil spill cleanup (discussed fully in Chapter 7).

2.2.1 Density, Specific Gravity and API Gravity

Density is the mass of a given volume of oil, variously expressed in units of g/mL or g/cm^3 or kg/m^3. Two measures of density are commonly applied to petroleum: specific gravity and API (American Petroleum Institute) gravity. Specific gravity is the density of a substance compared to that of water (and as a ratio is dimensionless). Because most oils are lighter than water, they float on it. For example, fresh water has a density of 1.0 g/mL and sea water a density of 1.03 g/mL at 4 °C. Therefore, an oil of specific gravity <1.0 (e.g., gasoline at 0.8) will float on water at that temperature and one with specific gravity >1.0 will

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sink at that temperature. As the temperature increases, the density (and thus the specific gravity) of oil decreases (Nmegbu 2014) . Thus, bitumen and certain heavy fuel oils may have densities greater than water at some temperatures and may submerge (Section 2.4.2.3).

API gravity (measured in degrees; ° API) , devised by the American Petroleum Institute, is an inverse measure of density that is used to compare different petroleum types (**Table 2.2; Figure 2.1**; Appendix C, **Table C1**). Fresh water is assigned a gravity of 10° API at 15 °C; 'light' crude oils have API gravity of >31.1°; 'medium' crudes are 22.3-31.1° API; 'heavy' crudes <22.3° API (or, more stringently, 10-15°); and bitumens 5-10° API (Speight 2014). API gravity can also be considered a rough indicator of crude oil quality, since more valuable light oils have higher °API values.

Oils of API >10° will float on fresh water at 15 °C whereas bitumen, defined as <10° API (Speight 2014), will sink. However, the sinking or floating of oil on water is additionally affected by salinity (e.g., seawater density 1.03 g/ml), by temperature and by interactions with particles (Section 2.4.2.3). Weathering of spilled oil, particularly diluted bitumen, can have a profound effect on the oil's density because evaporation (Section 2.4.1.2) of the LMW components ('light ends') leaves residual oil of higher density that may no longer be buoyant. Thus, the decrease in specific gravity (increase in °API) of spilled oil at higher temperatures may gradually be overcome by faster evaporation rates, resulting in an oil that is more dense.

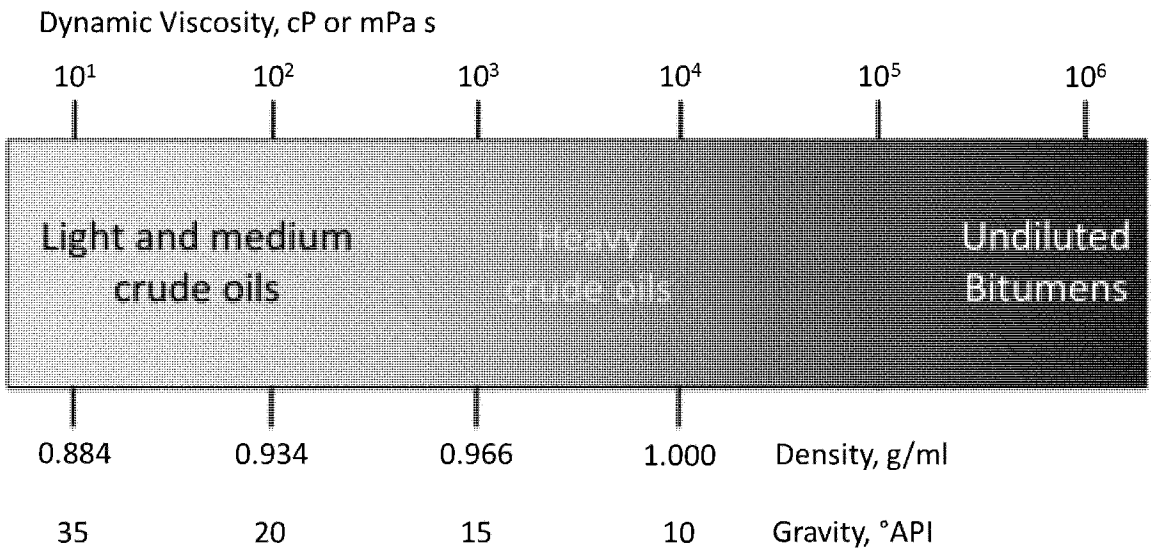

Figure 2.1 Relationship among viscosity, density and API gravity over a range of petroleum types. *Adapted from Speight (2014).*

2.2.2 Viscosity and Pour Point

Viscosity, defined as resistance of a liquid to deformation by shear or flow (WSP 2014), is also informally described as resistance to flow, or the 'thickness' of a fluid. The lower the viscosity, the more easily it flows. Two types of viscosity may be reported: 1) dynamic viscosity (resistance to shear, where adjacent layers move parallel to each other but at different speeds), expressed as centipoise (cP) or milliPascal second (mPa s); and 2) kinematic viscosity (the ratio of the dynamic viscosity to the

> Oil viscosity, like density, is determined in large part by its chemical composition. Higher proportions of 'light' components, such as LMW alkanes and aromatics, contribute to lower viscosity; whereas, heavier components, such as asphaltenes and resins, increase petroleum viscosity.

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density of the fluid) reported in centiStokes (cSt) or m^2/s. The significance of viscosity in oil spill response is discussed in Chapter 6.

Gasoline has a dynamic viscosity of 0.5 mPa s, light crude viscosity may be 5 -50 mPa s, whereas 'conventional' crude oils have viscosity up to 10,000 mPa s (similar to that of molasses) and 'ultraheavy' oils and bitumen usually have viscosity >50,000–100,000 mPa s (Speight 2014), similar to peanut butter. The relationships between viscosity, density and API gravity are shown in **Figure 2.1**. Increased viscosity decreases oil spreading on water (Section 2.4.1.1), dispersion (formation of oil droplets in water; Section 2.4.2.2) and the efficacy of chemical dispersants during remediation (Chapter 6).

As shown in **Figure 2.2**, kinematic viscosity is not only a function of the average molecular weight of the oil, it also declines exponentially as temperature increases (Nmegbu 2014). Thus, temperature impacts the extent of oil spreading on surfaces and penetration of shoreline sediments. Temperature also defines the oil's pour point - the temperature below which an oil will not flow but rather starts to solidify or gel. Bitumen, for example, does not flow in the reservoir and must be heated or, alternatively, bitumen must be diluted with light oils for transport (Section 2.3.7). The interrelatedness of viscosity, pour point and temperature then become s particularly important when considering the release of heavy oil or diluted bitumen from a heated pipeline or flow of an oil spilled onto ice or cold water.

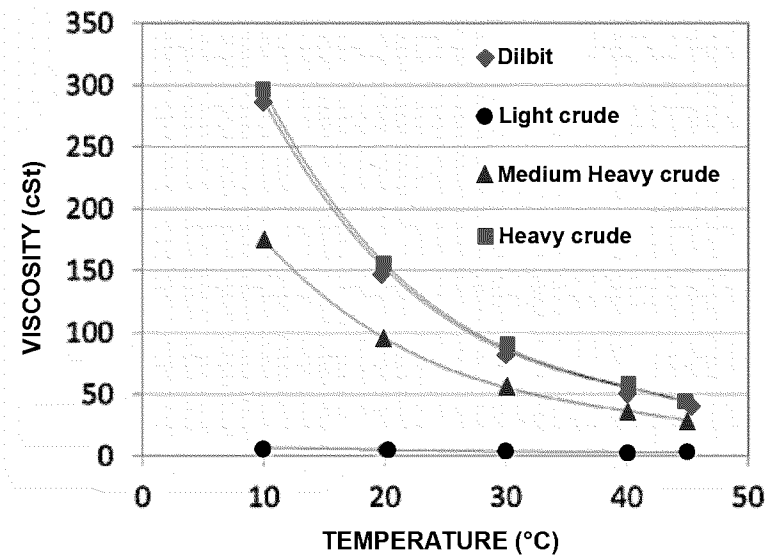

Figure 2.2 Relationship between temperature and kinematic viscosity *(reported in centiStokes [cSt]) for four oil types (Table 2.2). Adapted from Tsaprailis (2014)*

2.2.3 Volatility and Flash Point

The 'light' components of petroleum that contribute to decreased viscosity and density (e.g., LMW saturates and aromatics) also tend to evaporate (volatilize) readily. Volatility is particularly important when considering diluted bitumen blends, which contain substantial proportions (≥30%) of light petroleum products that pose a flammability hazard, even though the majority of the oil mass (the bitumen) is considered non-flammable.

High concentrations of light , volatile hydrocarbons contribute to the flammability potential of an oil by lowering the flash point, the lowest temperature the fuel must experience to achieve ignition in the air. Notably, these light hydrocarbons, particularly the monoaromatics, also correlate with ac ute toxicity to life forms (as discussed in Chapter 4). Thus, vapours from an oil spill can pose a hazard to spill responders, as well as to wildlife exposed to the fumes.

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After volatilization of some or most of the light components through weathering processes in open conditions (Section 2.4), the oils will have decreased flammability and acute aqueous toxicity. Temperature is obviously a primary factor affecting volatility and flash point, with higher temperature speeding volatility losses from spilled oil, but the thickness of the oil layer (which will also respond to temperature via viscosity changes) will also affect the rate of evaporation.

2.2.4 Water Solubility

In general, the saturates are highly water-insoluble compared with aromatics of a similar carbon number (e.g., the aqueous solubility of naphthalene, a C_{10} PAH, is approximately 600-fold greater than the C_{10} saturate *n*-decane). Monoaromatics are the most water-soluble of the hydrocarbons, whereas (alkyl-) PAHs are poorly soluble (e.g., at parts-per-billion concentrations) yet retain their toxic properties. For example, the aqueous solubility of benzene (**Table 2.1**), the most soluble BTEX compound, is 1.8 g/L water at 15 °C; solubility of *n*-hexane is 0.0013 g/L at 20 °C; naphthalene is 0.031 g/L at 25 °C; and *n*-hexadecane (C_{16}) and cyclohexane (C_6) (**Table 2.1**) are considered water-insoluble. Small aromatic heterocycles and resins, being polar, may be more soluble than their hydrocarbon counterparts. For example, whereas the hydrocarbon benzofuran (**Table 2.1**) is considered water-insoluble, its S-containing analog benzothiophene is soluble at 0.13 g/L at 25 °C and indole, its N-containing analog, is soluble at 3.6 g/L at 25 °C. However, large resin compounds and asphaltenes, despite their polarity, are water insoluble and the resins in particular tend to partition to the interface between oil and water.

> Most petroleum hydrocarbons are virtually insoluble in water. Only the LMW aromatics (e.g., BTEX), very small saturates and small polar compounds (some resins) have appreciable solubility in water, typically in the range of parts-per-million (ppm). Water solubility makes these molecules potentially more toxic because they are more 'bioavailable' to aquatic life if they subsequently partition into biological membranes.

High molecular weight components of bitumen, such as asphaltenes and most resins, are thought not to be toxic because they have extremely low solubility in water and do not readily cross biological membranes. However, diluents added to bitumens contain LMW compounds that can dissolve in water and contribute to acute aqueous toxicity before they are lost by weathering. In addition, heterocycles (e.g., dibenzothiophenes; **Table 2.1**) and 3-to-5-ringed alkyl PAHs detected in dilbit are derived from bitumen. Their concentrations in dilbit are similar to those of conventional crude oils and are thought to be the components causing chronic toxicity to fish embryos (Madison et al. 2015); see Chapter 4.

A measure of water solubility that is particularly pertinent to biological effects of petroleum is the octanol-water partition coefficient, which describes the concentration of a chemical in the solvent octanol ($C_8H_{15}OH$, a mimic for biological lipid membranes) versus its concentration dissolved in water, in a two-phase system at equilibrium for a given temperature. This ratio is usually expressed as the logarithm of the ratio (i.e., as log K_{ow}). The greater the K_{ow} value, the less polar and less water-soluble the chemical. Thus, K_{ow} helps predict the propensity of a petroleum constituent to dissolve into water and subsequently partition into a biological lipid membrane, an important factor in toxicity (Sikkema et al. 1995). K_{ow} and toxicity are discussed in detail in Chapter 4.

2.2.5 Surface Tension and Interfacial Tension

At liquid-air interfaces, surface tension is the measure of attraction (cohesion) between the surface molecules of a liquid rather than to molecules in air (adhesion). Information on the surface tension of different crude oils at various temperatures is available in Environment Canada's Oil Properties Database (http://www.etc-cte.ec.gc.ca/databases/oilproperties/). In the case of two immiscible liquids such as oil and water, the force between their surfaces is referred to as interfacial tension. The higher the interfacial

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tension, the less the oil will spread on water; whereas, if the interfacial tension is low, the oil will spread evenly without help from wind and water currents. Because higher temperature reduces interfacial tension, oil is more likely to spread in warmer waters than in cold waters. However, interfacial tension has less effect on oil spreading than viscosity.

2.2.6 Adhesion

Oil adhesion is the property of petroleum sticking to a surface, whether it is the rock formation comprising the oil reservoir, the production equipment or environmental surfaces after a spill. It is not included in the standard suite of oil properties reported by petroleum producers and is rarely discussed in the oil spill literature, but might be a predictor of oil spill cleanup needs if further measured and calibrated against a variety of oil types and their behaviour in the environment. A semi-quantitative gravimetric method has been proposed (Jokuty et al. 1995) in which a stainless steel needle is dipped in oil that is then allowed to drain, and the mass of oil adhering to the needle after 30 minutes of draining is weighed to produce a measurement with units of mass per surface area (g/m^2). However, the method has been neither adopted as a standard nor vetted. The relationship between evaporative weathering and adhesion is discussed in Section 2.4.1.2.

2.3 Examples of Oil Types

This section introduces a selection of oil types that are relevant to petroleum transport in Canada (**Table 2.2**) and/or to this Report. Within each oil type, one or more specific crude oils or blends is presented to highlight the differences between oils in terms of their chemical and physical properties, known or predicted behaviour and fate in the environment, timelines of weathering processes, and expected residues and spill response strategies (discussed in detail in later chapters).

With few exceptions this Report does not consider refined products, such as jet fuel, gasoline or diesel, nor liquefied natural gas or biofuels. The exceptions include products used to dilute bitumen for transportation and selected refined heavy fuel oils, for comparative purposes. The information presented in **Table 2.2**, and Appendix C (**Table C1**) has been collated from many different sources and is representative but not comprehensive. Two major online sources of information for oils transported in Canada are the Environment Canada Oil Properties Database (*http://www.etc-cte.ec.gc.ca/databases/oilproperties/*) and the Crude Quality Inc. industry website (*crudemonitor.ca*). However, both are incomplete in scope and compositional information. The Environment Canada website hosts a valuable catalogue of various chemical and physical data for 450 oils produced and transported globally but, regrettably, it lacks entries for many of the emerging unconventional oil types transported in Canada (e.g., dilbit, synbit, shale oil). The *crudemonitor.ca* website provides monthly analyses of oils currently transported inter-provincially in Canada, including unconventional oils, but publishes chemical composition data only for the light ends ($\leq C_8$) and general parameters, such as TAN, metals and sulphur content (and High Temperature Simulated Distillation curves [HTSD; Appendix A]) for heavy oils and blends. These data are relevant for achieving pipeline transportation specifications but not necessarily for oil spill response. Full spectrum analyses of heavier components that are predominant in heavy oils and bitumens and highly relevant to environmental cleanup are not available on this website.

Recommendation: Online public databases with compositions and properties of oils transported in Canada should be expanded to provide support for oil spill preparedness, response decisions and for the safety of first responders to a spill. To enable collection of relevant chemical data, expanded and standardized analytical methods must be developed, particularly for characterizing bitumen blends, such as dilbit and synbit.

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Table 2.2 Properties of some oil types commonly transported in Canada and/or relevant to this Report.[a]

Oil type and examples	Region of origin	Relevance
Natural condensates and refined products used as diluents for transport		
Sable Island condensate	Nova Scotia	40° API; Naturally occurring sweet ultra-light crude oil, often produced in conjunction with natural gas; predominantly volatile hydrocarbons
Fort Saskatchewan condensate blend	Alberta	68° API; Refined products, predominantly LMW aliphatics (C_5-C_6), with minor BTEX; used as diluent for oil sands processing and for bitumen transport
Cold Lake Diluent	Alberta	69° API; Natural gas condensate used as a bitumen diluent
Naphtha	Alberta	57° API; Refined product, predominantly $\leq C_{12}$ saturates and monoaromatics used as diluent for oil sands processing and for bitumen transport
Southern Lights Condensate Diluent	US	80° API; US condensate shipped to Alberta; predominantly C_5-C_{12} saturates
Suncor Synthetic Crude Oil (SCO)	Alberta	33° API; Refined product derived from partially upgraded bitumen and used as diluent in synbit
Light crude oils		
Alberta Sweet Mixed Blend (ASMB) Reference #4	Alberta	36° API; Blended aggregate of light, sweet conventional crude oils shipped from Alberta; used as a benchmark and laboratory reference oil by Environment Canada
Macondo (MC252)	Gulf of Mexico, USA	35° API; Oil released during DWH blowout; rich in LMW saturates that are gases or light liquids at atmospheric pressure
Norman Wells	Northwest Territories	38° API; Light crude from Canadian Arctic; wells drilled from artificial islands in Mackenzie River
Statfjord	Norway	38° API; North Sea offshore light oil
Arabian Light	Saudi Arabia	~32° API; Sample from St. John, NL, refinery
West Texas Intermediate (WTI)	Texas, USA	36-41° API; Benchmark oil for North American oil market pricing
Medium crude oils		
Alaska North Slope (ANS)	Alaska, USA	29° API; Conventional oil from Arctic; spilled from *Exxon Valdez* oil tanker into Prince William Sound, AK, in March 1989
Atkinson Point	Beaufort Sea, Canada	24° API; Oil from an onshore discovery well on Tuktoyaktuk Peninsula, NWT; no current production
Prudhoe Bay-A (PBCO)	Alaska, USA	29° API; Alaska North Slope crude; Used as reference oil by US-EPA in 1995

Oil type and examples	Region of origin	Relevance
Gullfaks and Troll	Norway	29° API; typical North Sea oils; used for oil spill research in Arctic regions e.g., Svalbard
Shale oils (Tight oils)		
Bakken	North Dakota and Montana (USA), Saskatchewan and Manitoba	42° API; unconventional light crude produced by fracking; may contain H_2S (23,000 ppm in an exceptional case); flammable agent in the 2013 LacMégantic QC, derailment and explosion
Waxy crude oils		
Hibernia and TerraNova	Newfoundland	34° API; Offshore Atlantic oils transported to mainland by tanker ship
Sour crude oils		
Midale	Saskatchewan, Manitoba	31° API; 2.4% S; Benchmark for Canadian medium sour crude oils
BC Light	British Columbia	41° API; sour light crude spilled into Pine River, BC, in 2000
Western Canadian Blend (WCB)	Alberta	22° API; 3.1%S; A blend of heavy sour crude oils, deliberately undiluted with butane that is co-produced with the crudes (to reduce explosion hazard) but instead diluted with condensate or light crude for transport
Heavy oils and diluted bitumen (See **Table 2.3** for definitions of various bitumen blends)		
Bunker C fuel oil(No. 6 Fuel oil) including HFO 7102	International	11° API; Common refined fuel oil class, often diluted with lighter petroleum for transportation; heavy component of oil spill in Lake Wabamun AB, 2005
Cold Lake Blend (CLB) dilbit	Alberta	23° API; Oil sands bitumen diluted with lighter petroleum; highest volume dilbit type transported in Canada; tested in wave tank trials for sinking and emulsification; one of the dilbits in the Kalamazoo, MI, oil spill in 2010
Albian heavy synthetic crude (AHS)	Alberta	20° API; Partially upgraded heavy crude oil, transported as dilsynbit;, used as refinery feedstock
Access Western Blend (AWB)	Alberta	23° API; sour heavy oil (3.9% S) produced by SAGD; tested in wave tank trials (Government of Canada 2013) for sinking and emulsification; composition of pipelined oil changes with season
Wabasca Heavy	Alberta	16° API; Heavy oil spilled near Mayflower, AR, in 2013
Western Canadian Select (WCS)	Alberta	19-22° API; Considered the benchmark for heavy, acidic (high TAN) crudes blended with bitumen streams to maintain TAN <1; diluted 10-50% with sweet synthetic crude oil and/or condensates; a

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Oil type and examples	Region of origin	Relevance
		component of the Kalamazoo, MI, dilbit spill in 2010
Synbit Blend	Alberta	21° API; Blend of several synbits plus heavy oils; evaporation behaviour differs from dilbit because of lower concentration of light ends in diluent
Athabasca/Cold Lake undiluted bitumen	Alberta	8-10° API; Density > 1.0; must be diluted or heated for transport
Orimulsion™	Venezuela	8° API; A proprietary emulsion of Orinoco bitumen with 26% fresh water and surfactant, density > 1.0; must be maintained above 30 °C to prevent gelling; limited production at present

[a]*Additional data including composition, specific gravity, sulphur content, adhesion and dispersability, where available, are given in Appendix C, Table C1. Note that the composition of several oils, particularly blended oils and dilbits, changes seasonally to maintain pipeline specifications for viscosity.*

2.3.1 Condensates and Refined Products Used as Diluents for Bitumen

Condensates are conventional naturally-occurring ultra-light crude oils, often co-produced with natural gas (methane). For example, Sable Island Condensate is produced from natural gas fields (e.g., Thebaud and Venture) off Nova Scotia and shipped to the mainland (Point Tupper, NS) by pipeline.

Condensates are rich in LMW saturates and, if spilled, would experience rapid evaporation, losing a high proportion of the spilled mass of oil. Because of the high saturate content, after evaporation and dissolution a weathered condensate would typically be highly biodegradable, given appropriate conditions, and little residual oil would be expected to persist in an open water spill.

Historically, natural condensate from western Canadian gas fields was used to dilute bitumen and heavy oils from the Athabasca and Cold Lake regions, but more recently the diluents are blends of light oils and/or refinery fractions, including naphtha and synthetic crude oil that is itself derived from bitumen (Segato [no date]; Crude Quality Inc. 2015). Although the diluents are predominantly composed of LMW materials far lighter than the bitumen, it should be noted that most diluents also contain appreciable proportions of the same HMW compounds found in bitumen that may not be detected by conventional GC (GC-FID or GC-MS), but are revealed using HTSD (Appendix A).

2.3.2 Light Crude Oils

Light crudes with API gravity >31.1 ° (Speight 2014) are conventional oils found throughout the world. Examples include Arabian Light, WTI (used as a pricing benchmark in the oil market), Statfjord from the North Sea and Norman Wells from northern Canada. Alberta Sweet Mixed Blend (ASMB), a major crude shipped from Alberta, has been used as a reference oil by Environment Canada for chemical composition, biodegradability and emulsification studies. Perhaps the most infamous light oil is Macondo (MC252) from the DWH oil well in the Gulf of Mexico that suffered a catastrophic blowout in 2010 (Section 2.4.2.1, Section 2.4.2.3 and Chapter 8), injecting not only liquid oil into deep seawater, but also light saturates that are gases at atmospheric pressure (methane, ethane, propane and butane).

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2.3.3 *Medium Crude Oils*

Oils between 22 -31° API are considered 'medium' crudes (Speight 2014). These conventional crudes include many of the ANS oils, Prudhoe Bay crude oil from the Arctic, and Norwegian North Sea oils produced from offshore wells. The latter oils have been used for research projects in the Arctic to determine the behaviour and fate of oil spills onto Arctic beaches. ANS oil was spilled in the highly publicized 1989 grounding of the *Exxon Valdez* in Prince William Sound, AK (Chapter 8), where up to 25% of spilled oil mass was lost by weathering and another ~30% was estimated by some researchers to have biodegraded within a few weeks of the spill (Atlas and Hazen 2011).

2.3.4 *Shale Oils (Tight Oil or Light Tight Oil)*

Shale oils [3] are unconventional liquid crudes that have only recently entered the North American oil markets in substantial volumes. As opposed to being newly 'discovered', many shale oil reservoirs were already known but were inaccessible pending recent technological advances afforded by horizontal drilling and hydrofracturing ('fracking') to release liquid oil that otherwise cannot flow because of 'tight' pores in the formation. One large known reserve is the Bakken formation straddling North Dakota, Montana and parts of southern Saskatchewan and Manitoba. Although Bakken oil is generally considered 'sweet' (i.e., low organic S and low H_2S), occasional samples have had high H_2S concentrations (American Fuel & Petrochemical Manufacturers 2014). Bakken shale oil was the flammable product in the Lac Mégantic, QC, rail disaster in 2013 and H_2S was implicated as a possible contributor to the explosion.

2.3.5 *Waxy Crude Oils*

These crudes contain the typical suite of petroleum fractions but also significant proportions of waxes (HMW linear paraffins, typically $>C_{20}$) and large naphthenes (*cyclo*-alkanes; **Table 2.1**). When present at high concentrations in petroleum, waxes may crystallize ('freeze') and form micro- or macroscopic deposits in the reservoir or infrastructure, depending on temperature and pressure. Because HMW saturates are less biodegradable than those of $\leq C_{18}$, spills of waxy oils may persist in the environment longer than non-waxy oils, although the wax residues have low chemical toxicity to aquatic life. Oils produced from the Hibernia and Terra Nova fields off the coast of Newfoundland are considered waxy crude oils.

2.3.6 *Sour Crude Oils*

A crude is considered 'sour' if it has more than 0.5 wt% total S content (>1% in some markets), as opposed to 'sweet crude' that has <0.5 wt% S. Most S in petroleum is included within aromatic or saturated structures in resins or asphaltenes (e.g., in heterocycles; **Table 2.1**) or is S^0, in which case it is not considered hazardous but is removed in the refinery at considerable expense. However, where oils contain highly toxic and potentially explosive H_2S gas, the H_2S concentration is reduced to safe concentrations before transport. The amount of H_2S varies considerably with the crude oil source (Hess 2012). Thus, both conventional and unconventional oils can be 'sour', regardless of the concentration of H_2S, with the unconventional oils more likely to contain organic S compounds that may resist biodegradation. Examples include the conventional medium sour crude Midale, and the WCB and AWB dibits; the latter was a component of the dilbit spill in Kalamazoo, MI (Chapter 8).

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[3] *Shale oil ('tight oil') should not be confused with oil shale. The latter is a sedimentary rock that harbours kerogen, a solid immature organic material that can be thermally processed at surface facilities into a shale oil that differs from the natural shale oil.*

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2.3.7 *Heavy Oils, Bitumen and Diluted Bitumen Products*

Heavy oils are defined as being <22° API or, more stringently, 10-15° API, with bitumens being semi-solids having density >1.0 g/cm^3 or 5-10° API (Speight 2014). Some heavy oils are refinery products and represent the 'heavy ends' that remain after distillation of lighter fractions like gasoline and kerosene from lighter oils. Bunker C fuel oil (also called No. 6 fuel oil or residual fuel oil; **Table 2.2**) is one such refined product that can be blended with lighter fractions to make an oil for combustion or used undiluted to make asphalt.

> Heavy oils and bitumen are at the end of a continuum of increasingly heavy, viscous oils with increasing proportions of HMW resins and asphaltenes and decreasing proportions of light molecules.

The distinction between heavy oil and bitumen is blurred by their similar chemical and physical properties and obscured by the working definitions and colloquial terms used by industry. Thus, different entities may assign the same oils to different categories. As noted by Winter and Haddad (2014), Exxon and *CrudeMonitor.ca* both identified the Wabasca Heavy Crude that spilled in Mayflower, AR, in 2013 (Chapter 1) as a 'diluted heavy crude', but the Canadian Government, Battelle (acting for API) and Penspen (acting for Canadian Energy Pipelines Association, CEPA) initially labeled it a 'diluted bitumen' (later corrected). The WCS blend that spilled in Kalamazoo MI in 2013 (Chapter 8) was called 'heavy oil' by Enbridge, but 'dilbit' by environmental agencies, such as Environment Canada (Government of Canada 2013). According to Yang et al. (2011), bitumen products have unique chemical markers that distinguish them from conventional crude oils. These fingerprints include the distribution of PAH, which is generally skewed towards the smaller multi-ringed structures (e.g., naphthalenes; **Table 2.1**) in conventional crudes versus larger PAHs (e.g., chrysenes) in bitumens, and have different distribution profiles within the alkyl PAH isomer series. However, in practice the two types of oil may be conflated in definition.

Regardless of nomenclature, it is generally accepted that both heavy oil and bitumen represent former conventional crude oils that have been extensively biodegraded and thermally altered *in situ* over geological time during uplifting or migration of the petroleum (Strausz et al. 2010; Fustic et al. 2012). Biodegradation depleted the lighter saturate fraction of the crude oil first, leaving poorly-degradable, complex, heavy molecules as residues. In addition, the products of incomplete microbial oxidation over geological time likely contributed to an increased resins fraction, yielding an extremely viscous, heavy petroleum (bitumen and extra-heavy oil) that cannot flow to production wells under normal reservoir conditions. This concept of liquid oil biodegradation to semi-solid bitumen over geological time is supported by the chemical compositions of heavy oils and bitumen, which have small proportions of light compounds known to be biodegradable (such as *n*-alkanes, *iso*-alkanes, BTEX and LMW PAH) and correspondingly higher proportions of poorly or non-biodegradable components (resins, asphaltenes and, of course, biomarkers like hopanoids, steroids and diamondoids; **Table 2.1**). Canada and Venezuela have the world's largest deposits of these highly viscous extra-heavy oils. However, Venezuela's Orinoco Belt, which is sometimes described as oil sands, has lower viscosity than the Canadian oil sands bitumen and can be transported as a 70:30 oil:freshwater emulsion (Orimulsion®; McGowan 1990) rather than being heated or diluted with light hydrocarbons. Whether natural or refined, extra-heavy oils contain much less biodegradable material than lighter crude oils.

There is a variety of diluted bitumen blends (**Table 2.3**). The most common diluents for bitumen transport include condensate (ultra-light oils partitioned from natural gas wells) or naphtha (a refined product), often at a ratio of 30% diluent to 70% bitumen for transport by pipeline (Crosby et al. 2013; Dew et al. 2015), whereas railbit for transport by rail tanker car is diluted half as much (i.e., ~15% diluent and 85% bitumen; Fingas 2014-2015). Synbit is a blend of bitumen and synthetic crude oil, a product of partial upgrading of bitumen that is heavier than condensate, at a ratio of approximately 50:50 (Crosby et al.

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2013). Synbit has different properties than dilbit because of the heavier nature of the diluent. Dilsynbit is bitumen diluted with synthetic crude oil plus diluent.

> Whereas some heavy oils may be suitable for shipping without alteration, bitumen does not flow unless it is heated and/or diluted. Therefore, bitumen is diluted with various light petroleum products to make it less viscous for different types of transport (rail, ship, pipeline).

In addition to season al changes in dilbit blends, the composition and proportions of diluents may differ in blends transported by inter- versus intra-provincial pipelines. This is because producers must conform to regulations when shipping oil outside the province, but may use specific blends to move bitumen between their own facilities. Notably, although the *CrudeMonitor.ca* database provides information on oils that are transported inter-provincially, it does not cover intra-provincial pipelines or those internal to an operator's facilities, which may have different specifications. For example, a paraffinic nC_5-nC_6 solvent used for bitumen extraction from oil sands ores by Shell Albian Sands in northern Alberta is also used as a diluent to pipeline the bitumen to the upgrading facility near Edmonton, but the solvent is too expensive to use as diluent for shipping to the US; instead it is recovered and returned by pipeline to the extraction facility (NRC 2013). Another example is the bitumen:water emulsion that spilled from an intra-facility pipeline at Nexen's Long Lake operation in July 2015 (Chapter 8). It is also important, as noted previously, to be aware of the limited petroleum characterization published by Crude Quality Inc. (*crudemonitor.ca*), specifically alkanes $\leq C_6$ and BTEX contents. These light hydrocarbons are important for safety and viscosity reasons during transport, but represent only a small proportion of heavy oils and bitumen blends in which the predominant heavy components are more relevant to oil behaviour and post-spill cleanup. Thus, there is a need for more internet-accessible oil composition data for emerging and unconventional oils to assist with planning, preparedness and response to spills.

Table 2.3 Definitions of diluted bitumen products *. Data from Fingas (2014-2015; 2015d), Dew et al. (2015) and Crude Quality Inc. (2015)*

Product	Description
Bitumen, Neatbit	Undiluted extremely heavy oil extracted from oil sands. Must be heated to be shipped
Diluent	Any light petroleum used to dilute bitumen for transportation by pipeline or rail; traditionally a condensate or ultra-light crude oil but now often a refinery cut such as naphtha [a]
Synthetic crude oil (SCO)	A liquid product made by partial upgrading or refining of bitumen; used as a diluent in synbit
Dilbit	Bitumen diluted with ~30% diluent, such as condensate or naphtha, for pipeline transportation
Railbit	Bitumen diluted with ~15% diluent (i.e., half as much diluent as dilbit), typically for transport by rail tank car
Synbit	Bitumen diluted ~50% with SCO
Dilsynbit	Bitumen diluted with SCO plus another diluent, usually a condensate (currently Albian Heavy Synthetic is the only dilsynbit transported)
Lightened Dilbit or C_4/C_5 enhanced Dilbit	Bitumen blended with a diluent supplemented with LMW alkanes such as C_4 (butane) and C_5 (pentane) [b]

[a] For some examples of condensate sources, see ISCO report #457, p. 8 (Fingas 2014-2015)
[b] Condensate is often in short supply in western Canada, in which case C_4 and C_5 may be used to replace a portion of the diluent while maintaining pipeline viscosity specifications. Potential problems with this supplement are that asphaltenes may precipitate during transport (Appendix A), and volatility and flammability increase (Section 2.2.3)

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Blends, such as dilbit and diluted Bunker C, may exhibit 'bimodal' properties that are non-linear with conventional crudes and therefore their behaviour can be difficult to predict based on chemistry alone. In fact, diluted bitumen may have the most variable composition of any transported petroleum product (Fingas 2015d) because of the composition and proportions of various diluents (natural condensates and ultra-light oils, butane-enhanced condensate, naphtha, synthetic crude, etc.) that are used to achieve pipeline transportation specifications of <300 mPa s and specific gravity <0.94 (Tsaprailis 2014; Fingas 2014-2015). Seasonal differences in diluent proportions and compositions (formulated to achieve regulated viscosity at ambient temperature) make chemical definition of diluted bitumens even more convoluted. This compositional variability gives the blends new properties that may not fully correlate with volume proportions and further confounds prediction of their behaviour in the environment.

2.4 Weathering of Oil Spilled in Aquatic Environments

Interactions of complex non-biological and biological processes lead to different fates of oil spilled in the environment. All of these are influenced by oil chemistry, environmental conditions and time, making prediction complicated and unique for each oil spill.

Weathering is a general term encompassing the changes in petroleum properties brought about by physical, chemical and biological processes when oil is exposed to environmental conditions such as in aquatic systems (Figure 2.3). These combined processes, along with the original oil chemistry and the time elapsed since the spill, affect the behaviour, fate, chemical composition and mass of residual oil after a spill event.

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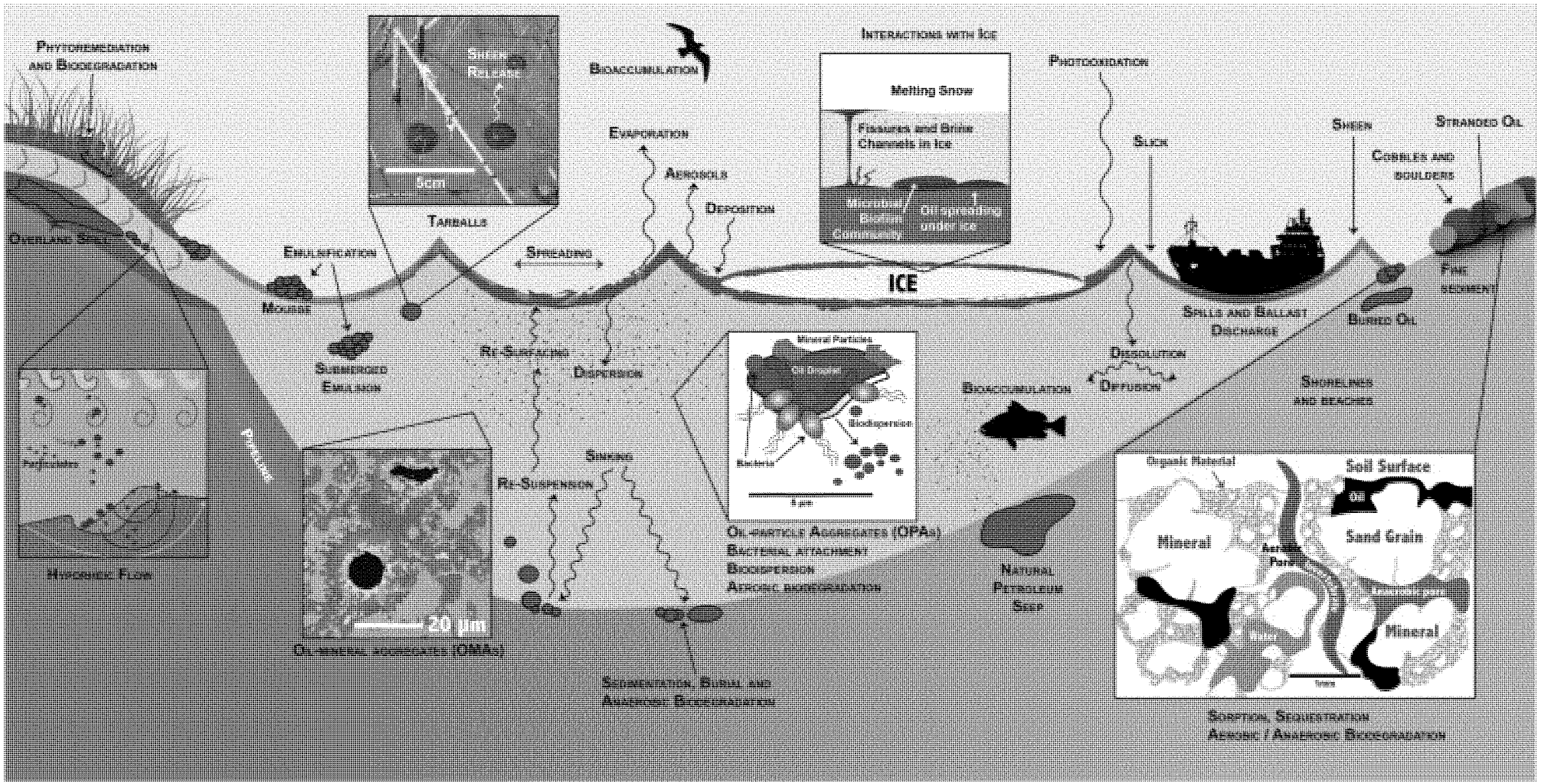

Figure 2.3 Overview of processes affecting the fate and behaviour of oil spilled in freshwater and marine environments. *The insets show processes at different scales. Adapted from Daling et al. (1990), Tonina and Buffington (2007, 2009), McGenity et al. (2012), AOSRT (2014), NRC (2014), WSP (2014) and Dew et al. (2015). Additional details on oil interactions with ice can be found in Figure 2.10.*

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Weathering processes occur at different rates and with different onset times, resulting in progressive changes in oil composition and behaviour after the spill (**Figure 2.4**). Some weathering processes, such as evaporation, begin immediately and are most significant in the short-term (within hours or days) ; others such as biodegradation occur after a delay or more slowly (over months or years). Therefore, gross weathering rates are not constant following a spill and are generally highest immediately after the spill. Moreover, weathering processes are not constant in all areas of a spill site. Oil at the surf ace of a waterbody will experience certain processes more severely than oil below the water surface, beneath ice, on the shoreline or at the edges of the spill compared to the thicker slick centre (as discussed below). As a corollary, certain environmental factors are key to the rate of weathering, such as temperature, wave action (energy), sunlight, suspended sediment in the water and microbial activity (as discussed in Chapters 3 and 6).

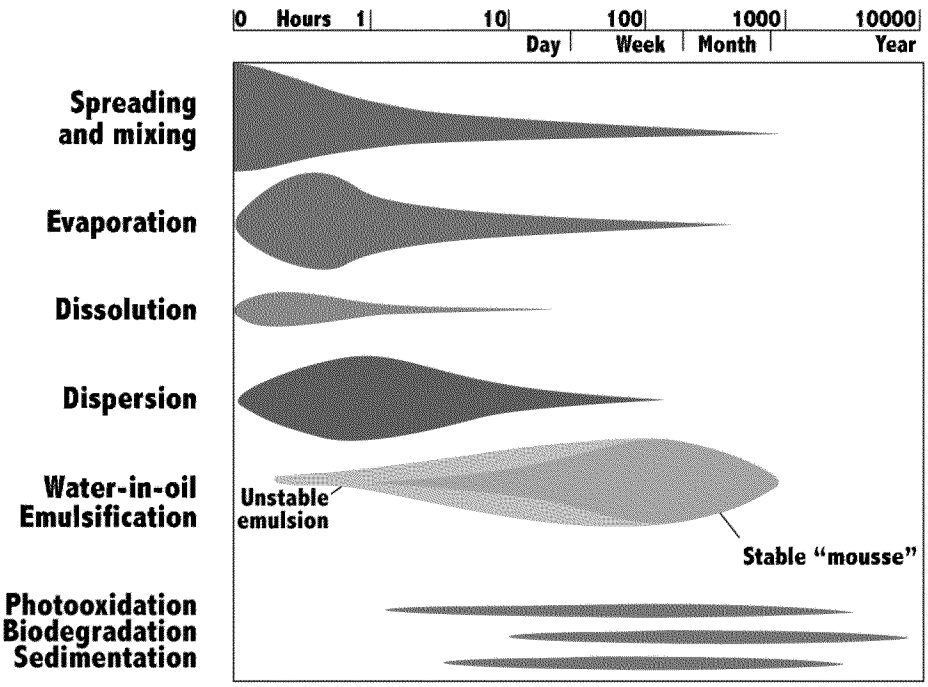

Figure 2.4. Time of onset and relative importance of weathering processes over time after an oil spill onto water. *The onset and magnitude of effect will vary with temperature and for different oils (note the time scale, which emphasizes the early onset of most processes). Figure adapted from AOSRT (2014).*

It is also i mportant to note that weathering is influenced greatly by the type of oil spilled. For example, light crude oils with high proportions of LMW hydrocarbons spread on water more readily and therefore are more susceptible to evaporation at the surface, whereas heavier oils have lower proportions of volatile hydrocarbons and are more likely to sorb to suspended sediments and subsequently sink (Section 2.4.2.3).

General short-term physical and chemical weathering effects on oil (**Table 2.4**) are organized below according to processes occurring at the water surface (fresh or marine water), in the water column, andalong the shoreline (riverbank or beach) or in underlying sediments. Notably, discussion of some processes is skewed by the availability of data for certain well -studied oil spills (e.g., DWH blowout and *Exxon Valdez* Oil Spill [EVOS]). Specific processes that depend on environment type and local facto rs at the impacted sites, including biodegradation and remedial intervention, are discussed in Chapters 3 and 6.

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Table 2.4. Effect of natural weathering processes on oil properties (NRC 2014).

Oil property	Natural weathering process	Effect on oil property
Viscosity	Loss of LMW components by evaporation and/or dissolution and/or biodegradation	Increased viscosity
	Formation of water-in-oil emulsions, including 'mousse'	Increased viscosity
Specific gravity	Loss of LMW components by evaporation and/or dissolution and/or biodegradation	Increased specific gravity
Volume of oil at surface	Loss of LMW components by evaporation and/or dissolution and/or biodegradation	Decreased volume
	Loss of small droplets by dispersion	Decreased volume
	Emulsification and/or biodispersion	Increased volume
Potential toxicity	Loss of LMW components by evaporation and/or dissolution and/or biodegradation	Decreased acute toxicity
	Formation of photooxidation products at surface, or of partially oxidized metabolites from incomplete biodegradation	Increased toxicity

2.4.1 Weathering Processes at the Water Surface

2.4.1.1 Spreading

When oil is spilled onto water and is allowed to spread unhindered, it moves away from the source where the oil layer is thicker (a 'slick') , forming a thin 'sheen' at the edges (**Figure 2.3**), e.g., the familiar 'gasoline rainbow' of 5 -10 □m thickness to nearly invisible sheens of <1 µm. As it thins, the slick may form patches or 'ribbons' (AOSRT 2014). Thus, spreading increases the spill area and decreases the average thickness of the oil layer. In the case of a sheen, the area of oil contamination can appear to be very large, but the actual mass of oil involved can be far smaller than in a slick. Spreading is influenced by the oil viscosity, water and air temperature, and wind, wave and/or current action; in turn, spreading affects evaporative losses (discussed below).

However, s preading is typically non -uniform, thicker 'windrows' (streaks of floating oil) separated by oil -free water or sheen are commonly observed as the spill progresses (Simecek-Beatty and Lehr 2007). These are understood to arise from mixing forces (Langmuir circulation) in the surface water (discussed in Chapter 5). The action of such vortices (Langmuir cells) producing patchy oil slicks will affect the efficiency of mechanical recovery of floating oil (Chapter 8).

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2.4.1.2 *Evaporation, Aerosolization and Atmospheric Re-deposition*

> Among the significant weathering processes occurring at the water surface, evaporation is usually the most immediate and rapid and has the greatest effect on the mass of spilled light and medium crude oils.

Evaporation can account for 75% of mass lost from condensates and ultra-light oils and 20-30% of losses from light oils, but ≤10% from heavy oils (NRC 2003), reflecting the different concentrations of volatile hydrocarbons in these different oil types. Saturates $<C_{15}$, monoaromatics (BTEX) and some small PAH are particularly subject to evaporative loss (AOSRT 2014) and 50% of hydrocarbons $\leq C_{16}$ were lost within 1 hour of an experimental oil spill by a combination of evaporation and dispersion/dissolution (Gros et al. 2014). The thickness of the oil layer also affects the rate of evaporation, with sheens evaporating more quickly than thick slicks and typically following an exponential curve (Fingas 2015e). Thus, evaporation can significantly reduce the total spill volume of a light oil. It also dynamically changes the composition of the surface oil which, subsequently, changes behaviour of the residual oil. For example, because volatile hydrocarbons, such as BTEX, tend to be the most acutely toxic, evaporation can reduce the acute toxicity of the spilled oil, but volatilization creates a potential explosion hazard above the spill and potential breathing hazards for oil spill responders. Simultaneously, with the loss of 'light ends', the residual oil becomes enriched in the HMW compounds and therefore becomes more viscous and dense, as well as becoming enriched in alkyl PAH that increase the chronic toxicity of the residual oil (Chapter 4). When the oil is unable to spread due to confinement or low wind/wave action, such as on a pond, a 'skin' of resins and asphaltenes can form on the surface, decreasing further evaporation of lighter components.

There is currently debate about the influence of surface winds over an oil spill. Fingas (2015c) pointed out that the air boundary layer above an oil spill theoretically can regulate the rate of evaporation, but that evaporative losses from light oils (e.g., ASMB, diesel, gasoline) in laboratory trials appeared to be limited more by slick thickness (affecting diffusion of volatile molecules to the oil surface) than by simulated wind speed. A previous report (Fingas 2004) indicated that temperature and time were greater factors in oil evaporation than surface wind velocity or oil layer thickness for a wide range of crude oils in which the bulk of hydrocarbons are $>C_{10}$. In contrast, Gros et al. (2014) determined that wind speed strongly influenced evaporation (as well as dispersion and dissolution) very early in an experimental spill of a Norwegian crude in the North Sea.

<u>Recommendation:</u> More information is needed for early evaporation and dissolution processes in actual oil spills to resolve present debate.

Whereas evaporation from a light or medium crude can be substantial (>30-50%), heavy oils experience far less evaporative loss. AWB and CLB dilbits lost only 15-18 % mass under quiescent conditions in an outdoor flume tank (King et al. 2014), and under laboratory conditions the latter lost a maximum of 17.4% mass at 22 °C (Waterman and García 2015). Synbit blends should experience even less evaporative loss than dilbits because the diluent (synthetic crude) has higher average molecular weight than the condensates or naphtha used in dilbit (Dew et al. 2015; Fingas 2015e). Diluted bitumen blends show bimodal behaviour, with light diluent components evaporating at a rate that decreases with time as the spilled oil becomes more viscous, slowing losses of volatile hydrocarbons. For example, elevated benzene levels measured in the air after dilbit spilled into the Kalamazoo River, MI (Crosby et al. 2013) (Chapter 3) represented evaporative losses from the diluent fraction. Dilbit that is initially buoyant (specific gravity <1.0) becomes denser (≥1.0) during evaporation, increasing the potential for sinking through the water column and subsequently contaminating sediments (Winter and Haddad 2014). Therefore, the

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early period of a diluted bitumen spill may be very important for efficient recovery and cleanup (Winter and Haddad 2014).

Yarranton et al. (2015) recently showed in laboratory tests that evaporation of diluent from Cold Lake Winter Blend dilbit was comparable regardless of whether it occurred from the surface of a glass slide or fresh water; i.e., contact with quiescent water had no additional effect on evaporative losses and, furthermore, air flow had little effect on evaporation rate. They determined that ambient temperature (5 °–25 °C) and dilbit film thickness (1.6–5.4 mm) were key parameters and that evaporation rate was limited by diffusion of the light diluent components through the dilbit film, which, in turn, correlates with changing density and viscosity of the product during weathering. In contrast, the evaporation rate of the light reference oil ASMB was limited by convective mass transfer of volatile components. This highlights the concern that existing evaporative models of dilbit spills may not adequately predict evaporative behaviour of dilbit, and that further testing using a range of diluted bitumen products is required for validation or refinement of evaporation models. Additionally, these laboratory observations highlight the potential for sinking of diluted bitumen products that reach densities >1 g/cm^3 by evaporative weathering, and may inform prediction of adhesion of the weathered product to surfaces or suspended sediments.

> Evaporative losses from dilbits are substantially less than from light crude oils, but have a greater effect on physical properties. As dilbit weathers, it exhibits bimodal behaviour as diluent volatilizes and bitumen dominates the chemistry of the weathered oil. Current questions include whether blending bitumen with a diluent yields a homogeneous fluid equivalent to a conventional heavy oil and, conversely, whether loss of diluent restores dilbit to the original bitumen composition and properties.

There is debate whether 100% of the diluent component of bitumen blends can be lost by natural evaporation (Fingas 2015e) or whether the residual bitumen/heavy oil will retain some of the diluent components as intimately blended constituents, conferring novel properties on the partially weathered oil (Winter and Haddad 2014). This is particularly important for predicting if weathered dilbit, synbit, etc., will float or sink in water (Section 2.4.2.3). It seems plausible that some higher molecular weight components of the diluent would be retained in the weathered oil, but at concentrations too low to significantly change the physical behaviour of the residual oil compared with the original bitumen or heavy oil stock. Although research has been initiated recently into the evaporative behaviour of various bitumen blends under actively mixed conditions (King et al. 2015a), these data are not yet published and further scrutiny is warranted. The observation of evaporative mass losses of <20% after rigorous weathering at environmentally-relevant temperatures for dilbits nominally comprising ≥30% diluent suggests that a substantial proportion of diluent remains intimately associated with bitumen.

The effect of evaporation on adhesion by dilbit is significant during the initial weathering phase when diluent is being lost at the fastest rate. For example, AWB dilbit showed a rapid increase in adhesion during the first 24 hours of natural weathering on seawater in open-air flume tanks (**Figure 2.5**). The effect on intermediate fuel oil (IFO 180) adhesion was less and after 24 hours there was little change in adhesion (or composition) of either oil.

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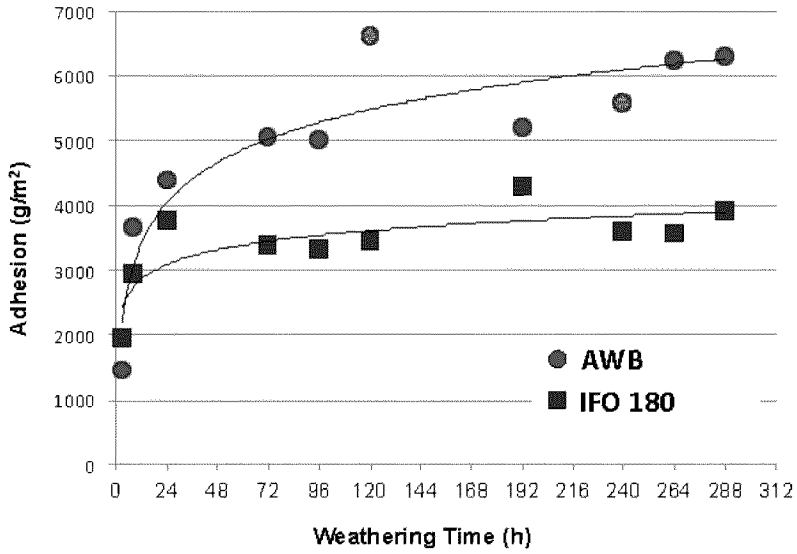

Figure 2.5 Effect of progressive natural weathering on adhesion. *Unpublished data for Access Western Blend (AWB) dilbit and an intermediate fuel oil (IFO 180) provided by Fisheries and Oceans Canada. Natural weathering was achieved in a n open-air seawater flume tank, and adhesion of the recovered oil was determined using the semi-quantitative method of Jokuty et al. (1995).*

Hollebone (2015) measured adhesion for several oils that experienced progressive evaporation in the laboratory. A range of semi -quantitative adhesion values for selected oils is given in Appendix C, **Table C1** and a comparison of adhesion by selected light, medium and waxy crudes, CLB winter blend dilbit and Bunker C fuel oil during controlled weathering is shown in **Figure 2.6**. Adhesion affected by evaporative loss follows a continuum, with near-linear responses by light crude oils having high proportions of saturates (Hollebone 2015) versus exponential responses by heavy oils with high resins and asphaltene contents, consistent with observations by Jokuty et al. (1995). As expected, the sol e dilbit sample showed bimodal behaviour, with little initial change in adhesion followed by an exponential increase similar to that of the heavy oils.

<u>Recommendation:</u> Further study on a range of oils under varied environmental conditions is needed to resolve the current uncertainty around evaporative losses experienced in surface spills, including the effect of oil layer thickness and surface winds, the role of photooxidation and formation of weathered 'skin'. Field work especially, but also lab work, are essential to determine the extent and kinetics of diluent evaporation from various bitumen blends, and the resultant behaviour of such blends due to weathering processes.

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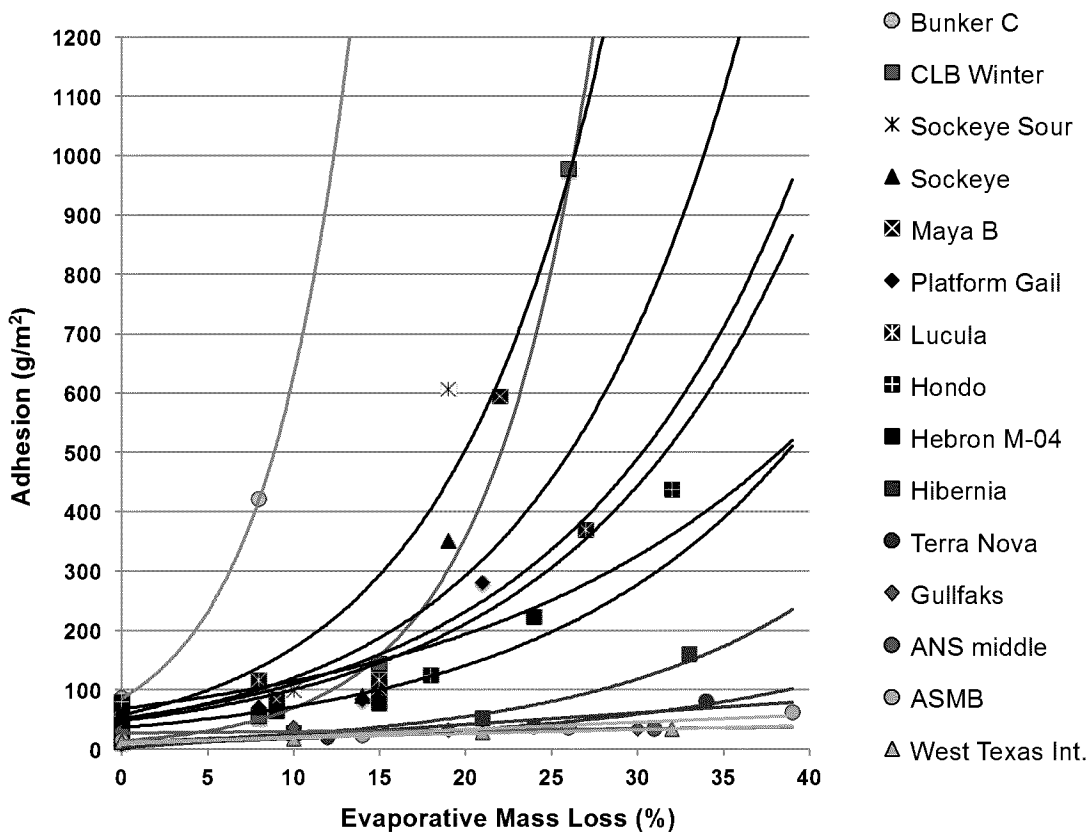

Figure 2.6 Correlation between adhesion and evaporation of selected oils. *Data compiled from Hollebone (2015) and unpublished data from Environment Canada (dilbit values); trend lines were drawn in Excel v.14.4.6. Evaporative loss was achieved using rotary evaporation in a laboratory and adhesio n was measured using the semi-quantitative gravimetric method of Jokuty et al. (1995). Blue symbols, light crude oils; red, medium crude oils; green, waxy crude oils; black, heavy crude oils; purple, dilbit (Cold Lake Blend-Winter); grey, Bunker C.*

Aerosolization is a recently proposed dispersal route for atmospheric transport of crude oil spilled onto water. Laboratory studies have shown that bubble bursting (e.g., due to ebullition of gases from a shallow subsea blowout) and/or wave action (e.g., simulated white caps) can create hydrocarbon aerosols by mechanisms similar to those that generate ocean spray (Ehrenhauser et al. 2014). Likewise, hydrocarbon aerosols are formed by the impact of raindrops on an oil slick (Murphy et al. 2015). In both cases, the presence of dispersant, such as Corexit 9500 used in the DWH spill (Section 2.4.2.1), enhanced the effect. Notably, this dispersion route could also significantly increase evaporation from air-borne droplets.

Atmospheric re -deposition of the aerosolized oi l (SL Ross 2012) onto water or land could distribute the hydrocarbons beyond the extent of the main spill. Anecdotally, during the 1984 Uniacke G-72 surface blowout off Sable Island, NS, which ejected a plume of condensate into the atmosphere, responders r eported 'hydrocarbon rain' near the spill site (K. Lee, pers. comm. 2015); a similar phenomenon occurred during the 1982 onshore blowout of a sour condensate well near Lodgepole, AB (J. Foght, pers. comm. 2015). However, literature on the magnitude and impacts of atmospheric hydrocarbons on water and land is sparse (NRC 2003), warranting further study.

2.4.1.3 Photooxidation

In a poorly understood free-radical-generating process, aromatic hydrocarbons (particularly PAHs and including aromatic N-, S- and O-heterocycles) react with oxygen in the presence of sunlight, yielding oxygenated products that are more water -soluble and usually more resistant to

biodegradation than the parent compounds. Whereas removal of PAH from the oil is beneficial, as some PAH s contribute to its potential carcinogenicity and embryotoxicity , the increased mobility and persistence of the photooxidized products in the water column are likely detrimental. The full suite of photooxidation products is currently unknown, although they are known to have impacts on aquatic life (Chapter 4). Notably, the susceptibility of PAH to photooxidation is the opposite of biodegradation potential: the smaller the PAH the more biodegradable and less susceptible to photooxidation; conversely, the more ar omatic rings, the less biodegradable but more susceptible to photooxidation.

It is likely that photooxidation is most important relatively early in the weathering process. Recent laboratory studies with fresh and weathered oils from the *Heibei Spirit* spill (a mixture of light and heavy crudes; Yim et al. 2012) used ultra -high resolution GC to show that photooxidation preferentially increased the proportion of oxygenated compounds in fresh oil versus weathered oil (Islam et al. 2005). The N -containing resins (**Table 2.1**), particularly compounds with secondary and tertiary amine groups, such as phenylamines, were most vulnerable to photooxidation. S-containing resins, such as aromatic dibenzothiophenes, were also photooxidized, with unknown environmental consequences for toxicity and pe rsistence of the weathered oil.

Photooxidation is obviously affected by sun angle and season, particularly at higher latitudes in the Arctic. However, even at lower latitudes the process appears to be slow, accounting for <0.1% of total losses per day (WSP 2014). Although photooxidation may not directly achieve significant mass losses, it can have indirect effects on subsequent weathering processes. For example, oxidized products may contribute to emulsification (Fingas 2015e) and crust or 'skin' formation on the surface of an oil slick (Bobra and Tenn yson 1989) that decreases evaporative losses, as discussed above. Unexpectedly, ultraviolet irradiation of light Macondo oil from the DWH oil spill affected formation of microbial flocs in seawater compared with fresh Macondo oil (Passow 2014). This 'photo -chemical aging' appears to have promoted subsequent sinking and sequestration of some of the oil in the Gulf of Mexico seabed (discussed in detail in Section 2.4.2.3).

It appears that there may be only a small window of time for oil spill response to redu ce the effects of photooxidation on other weathering processes, but further research into the products and magnitude of photooxidation is required, especially for emerging, unconventional oils and blends.

2.4.1.4 Emulsification

Formation of emulsions (**Figure 2.7**) is important for several reasons. By incorporating up to 60 - 80% water, stable emulsions increase the effective volume of the spilled oil up to two to five fold and increase the oil's viscosity by up to 1,000 -fold (Fingas and Fieldhouse 2015). This decreas es evaporation of volatile components and reduces oil spreading. Oil entrained in mousse emulsions resists dispersion with chemicals and biodegradation due to its viscosity, low bulk surface area and hindered nutrient replenishment (AOSRT 2014). Emulsions tend to move from floating on the water surface to being submerged in the water column, where physical recovery may be hampered. All of these behaviours affect cleanup and remediation response options (Chapter 6).

The type of emulsion formed , whether water -in-oil (w/o) or oil -in-water (o/w), depends on the environmental conditions (temperature, mixing) and on the mass and chemical composition of the oil. Some crude oils form w/o emulsions rapidly but soon revert to two discrete oil and water phases; others, such as heavy fuel oils , form w/o emulsions poorly or only slowly. Some w/o emulsions can be stable for months or years (Fingas and Fieldhouse 2015), likely stabilized by sub-fractions of asphaltenes (Yang et al. 2014, 2015), waxes acting synergistically with asphaltenes (Kokal 2002), resins (WSP 2014), or even by bacterial cells (Dorobantu et al. 2004)

that prevent coalescence of the oil by aligning at the oil:water interface (**Figure 2.7**). Eventually the emulsions may separate into oil and water again by natural processes , including additional weathering, oxidation and/or fr eeze-thaw action (Fingas 2014).

> When sufficient mechanical energy (e.g., wind and/or wave activity) is applied to mix oil and water, unstable or stable emulsions may be generated; the latter may take the form of oil droplets dispersed in the bulk water phase or, conversely, water droplets in a bulk oil phase commonly known as 'chocolate mousse'.

In light of the proposed roles of asphaltenes and resins in stabilizing emulsions, it is both interesting and unexpected that bitumen and severely weathered diluted bitumen that have a high content of both classes do not appear to form stable emulsions (Fingas and Fieldhouse 2015), although lightly w eathered bitumen blends can incorporate water droplets given sufficient mixing energy. During outdoor wave -tank tests where two weathered dilbit products (AWB and CLB, **Table 2.2**) were applied to seawater at an average temperature of 8 °C with and without a rtificial sediment (fine clay minerals), breaking waves caused the oil to form unstable w/o emulsions and large submerged droplets that readily coalesced into a surface slick (Government of Canada 2013). This observation was confirmed by another study of f resh and weathered Cold Lake dilbit and ASMB light oil (**Table 2.2**) agitated at 15 °C in an indoor wave tank with river water containing natural floodplain sediment (Zhou et al. 2015). The ASMB immediately dispersed into the water column whereas the dilbit formed an unstable emulsion that subsequently separated and remained floating for eight days or became stranded on the apparatus 'beach'; no formation of mousse was observed with the dilbit. Poor emulsion formation by heavy oils and weathered dilbits may o ccur because high viscosity limits dispersion and therefore the amount of water incorporated, and such water is only transiently entrained not actually emulsified. It is notable that this unanticipated behaviour was revealed only by experimentation rather than from first principles, highlighting the importance of research on unconventional oils. Further studies on the fundamental mechanisms of dilbit interactions with water are underway to help address this knowledge gap (e.g., Dettman and Irvine 2015), inc luding the effects of emulsification on adhesion properties (Section 2.2.6).

<u>Recommendation:</u> Laboratory and field trials are needed to determine the conditions under which fresh and weathered diluted bitumen blends will form stable or unstable emulsions in fresh water and seawater . A variety of bitumen blends would be essential to investigate under various agitation conditions and salinities to inform modeling and response to spills.

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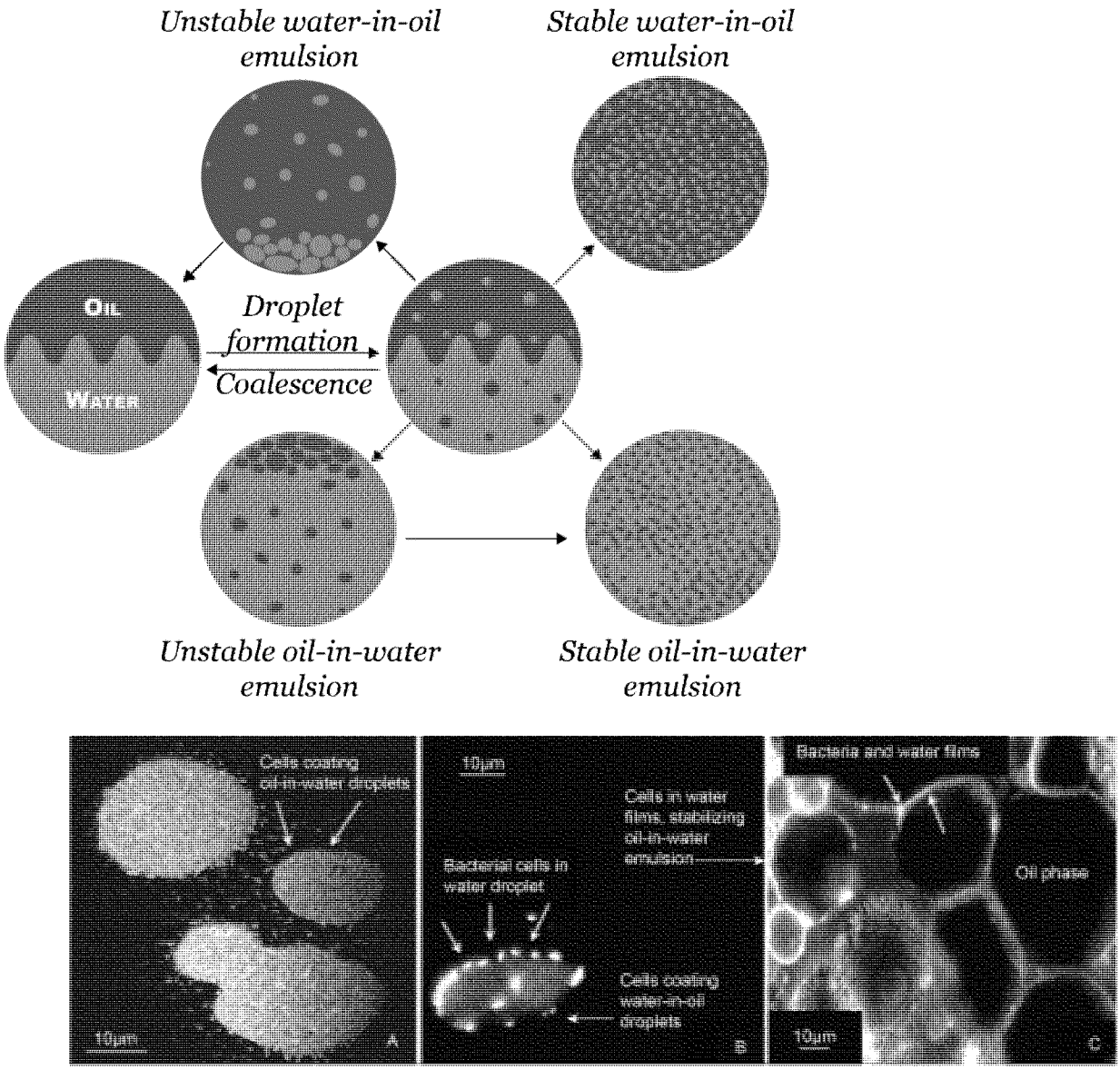

Figure 2.7 Emulsification of oil and water. *Upper panel: Generation of stable and unstable water-in-oil (w/o) and oil-in-water (o/w) emulsions in seawater. Lower panel: Stabilization of o/w and w/o emulsions in artificial fresh water by cells of hydrocarbon-degrading bacteria. Lower panel adapted from Dorobantu, L.S., Yeung, A.K.C., Foght, J.M. and M.R. Gray. 2004. Stabilization of oil-water emulsions by hydrophobic bacteria. Applied and Environmental Microbiology 70(10): 6333-6336. Copyright © American Society for Microbiology.*

2.4.2 Weathering Processes in the Water Column

2.4.2.1 Dissolution

Evaporation and dissolution are interrelated and competing processes. In the former process, petroleum constituents are moved to the atmosphere from the surface of the slick, and in the latter, they are diluted in the water column from the underside of the slick. Both processes reduce the potential acute toxicity of the residual oil while potentially increasing the chronic toxicity by enriching the remaining oil in alkyl PAH (Chapter 4). However, evaporation typically accounts for far greater mass losses of spilled oil than dissolution.

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Dissolution is important because many of the most water-soluble components of petroleum (e.g., BTEX) are also the most volatile and most acutely toxic.

Hydrocarbons typically have very low solubility in water (Section 2.2.4), which decreases with molecular size. Monoaromatics and the lightest saturates are the most water-soluble, whereas larger saturates and PAH have lower solubility, and asphaltenes (and most resins) are considered water-insoluble. For this reason, heavy oils and bitumen do not readily dissolve in aqueous solution (the water soluble fraction is <10 ppm) and dissolution does not significantly reduce the total mass of heavy oils spilled onto water. However, LMW components of the diluents in dilbit and synbit will have some aqueous solubility, in which case dissolution of even a small fraction of a very large dilbit spill may represent a large mass of dissolved chemical. Dispersion of the oil as small droplets with large ratios of surface area to volume would hasten and increase such dissolution.

The relative importance of dissolution increases in deep subsurface spills, such as the DWH blowout, where the long travel time for oil droplets to rise or travel laterally through the water column increases the opportunity for small molecules to dissolve rather than evaporate upon reaching the water surface (FISG 2010). Furthermore, subsurface application of the chemical dispersant Corexit 9500 to the DWH blowout (Chapter 6) decreased oil droplet size, increased the total oil:water surface area to facilitate partitioning into the water phase, generated a subsurface plume of droplets <100 µm diameter that moved horizontally with deep currents in the Gulf of Mexico, and prolonged oil contact with the water (Thibodeaux et al. 2011). This 'continuous molecular extraction' of oil droplets by the water column, which did not reach equilibrium, preferentially depleted the oil of the most soluble monoaromatics, some resins and even PAHs, alkyl PAHs and light alkanes $<C_8$, and enriched the alkanes $\geq C_9$ and heavier compounds in the oil droplets. This phenomenon may have decreased the acute toxicity of dispersed Macondo oil and thereby increased its biodegradability (Section 2.4.2.5 and Chapter 6).

<u>Recommendation</u>: There is a need for the development of new dynamic (non-steady state) models describing dissolution of hydrocarbons in deep seawater and turbulent rivers, especially for diluted bitumens where the fate of such hydrocarbons is not yet understood. Similarly, toxicity testing of different water-soluble fractions (with and without dispersant addition) is needed for unconventional oils and blends that exhibit bimodal behaviour.

2.4.2.2 Natural dispersion

Natural dispersion of oil in the water column (as opposed to addition of chemical dispersants as a cleanup intervention; Chapter 6) occurs when the mechanical action of waves or turbulence detaches oil droplets from the slick and forces them into the water column. Likewise, turbulent flow likely leads to oil dispersion in rivers having steep gradients, high-velocity flows and/or boulder/cobble substrates, and during offshore subsurface well blowouts like the DWH (Zhao et al. 2014). Depending upon droplet size, depth and energy of the system, droplets may remain dispersed (i.e., suspended in the water column) or may resurface with or without coalescing with other droplets (**Figure 2.8**). Droplets <20 µm (0.02 mm) may remain stably dispersed in water without resurfacing for a long time (Fingas 2014b), as observed with the DWH oil (Valentine et al. 2014).

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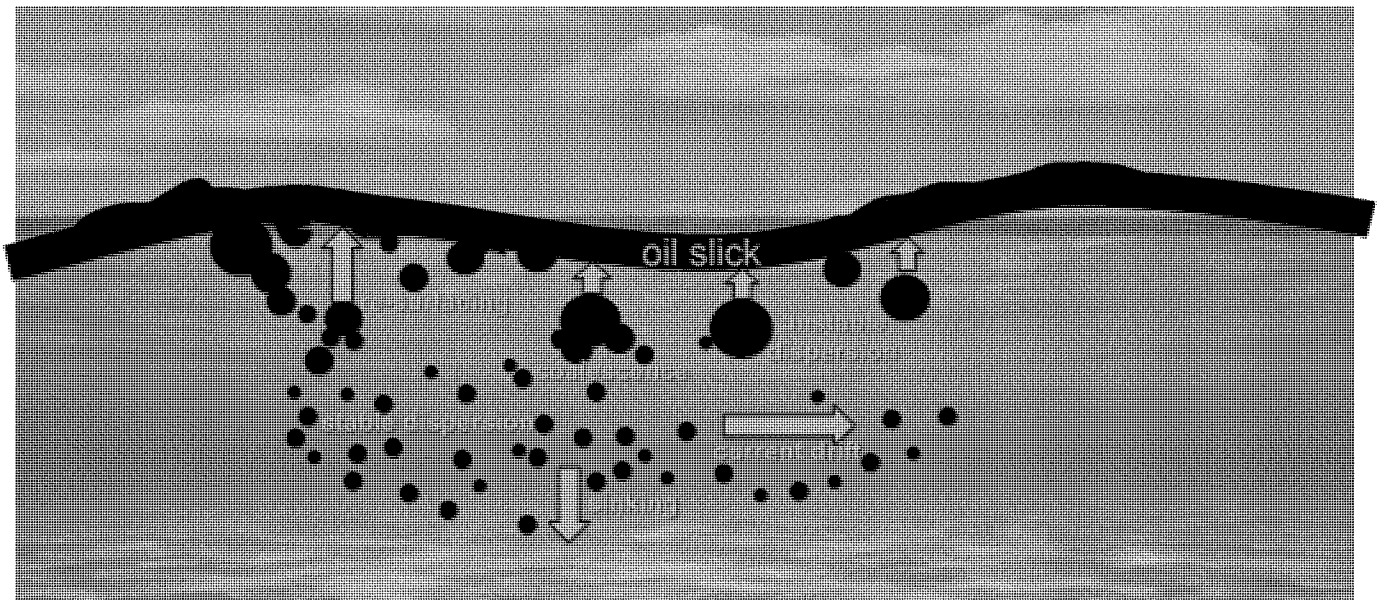

Figure 2.8 Natural dispersion of oil into the water column.

Dispersion affects oil behaviour in several ways. First, stable dispersion removes oil from the surface, reducing evaporation and photooxidation. Second, the greater surface area afforded by small oil droplets makes the oil more available for dissolution (Section 2.4.2.1) and biodegradation (Brakstad et al. 2015; Prince 2015; Section 2.4.2.5). In fact, the rate of oil biodegradation has been suggested to correlate with oil droplet size distribution (Venosa and Holder 2007). Moreover, many oil-degrading microbes are attracted and attach to the oil–water interface (Kang et al. 2008a,b; Abbasnezhad et al. 2011a,b) where they may: (a) produce biosurfactants to further increase droplet formation; and/or (b) stabilize oil droplets by preventing coalescence, as described for emulsions (Dorobantu et al. 2004; **Figure 2.7**). Oil droplet size is a factor in toxicity to aquatic organisms, as zooplankton may ingest tiny droplets and thus pass the hydrocarbon up the food chain, and smaller droplets (with a higher ratio of surface area to volume) facilitate dissolution of hydrocarbons associated with toxicity (**Figure 2.3** and Chapter 4).

Although dispersion of conventional crude oils is relatively well understood (see Daling et al. [2003]), considerably less is known about dispersion of bitumen blends in fresh water, estuarine (brackish) water or seawater (Dew et al. 2015). Recently, dispersion of CLB dilbit in filtered seawater was conducted in a wave tank under spring-time and summer temperatures (~8.5 °C and 17 °C, respectively), with or without addition of fine mineral particles (King et al. 2015b) and with or without chemical dispersant. With breaking waves, the dilbit showed poor natural dispersion (only 6% dispersion effectiveness) in the presence and absence of suspended fine mineral particulates. Addition of chemical dispersant alone efficiently produced droplets (45-59% dispersion effectiveness), but the droplet size distribution differed from that of chemically-dispersed conventional crude oils. Additional studies of dilbit dispersion are underway in Government of Canada laboratories, but data are not yet available for review.

<u>Recommendation:</u> More data are required for natural and chemically-enhanced dispersion of unconventional oils and diluted bitumen blends, especially in fresh water.

2.4.2.3 Submergence, sinking, sedimentation and re-suspension of oil

Multiple processes can alter oil density to influence submergence, sinking and sedimentation, including: increased density due to evaporation and/or dissolution of light components (Section 2.4.1.2 and Section 2.4.2.1); emulsification (Section 2.4.1.4); and interactions with particles. The

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latter process can be further delineated by considering inorganic particles (e.g., minerals and detritus suspended offshore, or onshore sediments if the oil spills over land before entering the water body) or organic particles (e.g., flocs of microbial biofilms, biological detritus and/or phytoplankton mucus).

> Oil may submerge through dispersion or emulsification and eventually sink through the water column to become entrained in the underlying sediments (marine or freshwater) in a process called 'sedimentation'. If disturbed, sunken oil may become re-suspended to enter the water column again, resurface and/or be re-distributed in the subsurface.

The density of many crude oils, including diluted bitumen products, is less than that of both fresh water (~1 g/cm ^{3}) and sea water (~1.03 g/cm ^{3}) and therefore these oils will float when spilled. However, bitumen by definition has a density >1 g/cm ^{3} and, if diluent evaporates from dilbit during a spill, the residual bitumen will become more dense and may become submerged or sink. Even so, oil droplets of density ~1.0 can be neutrally buoyant and remain suspended in the water column, especially if present in turbulent water such as in a rapidly flowing river or high wave energy system. In rivers or at beaches, suspended oil droplets could be entrained into porous gravel by hyporheic flows (**Figure 2.3)**, creating a potential for contamination of sediments and pore water with free-phase oil or dissolved hydrocarbons (reviewed in more detail in Chapters 3 and 5). In addition to evaporation, other environmental processes, such as photooxidation (Section 2.4.1.3), can affect oil buoyancy (NRC 1999; Government of Canada 2013). As water temperature generally decreases with water depth, the oil density will increase, possibly causing it to remain submerged or sink, and temperature cycling between day and night may cause oil to alternatively submerge and float. Emulsification can also cause floating oil to become submerged (but typically not to sink).

> OMAs form when oil interacts with and adheres to inorganic particles. A more general term, OPAs, includes OMAs plus oil associated with organic particles.

Interaction of oil with non-oil particulates changes the properties of the aggregates, causing normally buoyant oil to sink, for example, or liquid oil to become semi-solid. 'Oil-Mineral Aggregates' (OMAs) can form when oil interacts with inorganic materials, such as clay minerals suspended in the water column (Lee 2002), or with soils or shoreline sediments during overland flow into a waterbody. The term oil-particle aggregates' (OPAs) collectively encompasses OMAs as well as oil associated with organic material, such as detritus and living microbial cells (Fitzpatrick et al. 2015; Waterman and Garcia 2015), and combinations of these particles (**Figure 2.9**). OPAs may have neutral buoyancy and float at or below the water surface, or may sink and become entrained in sediments. The latter fate would potentially place the oil in anaerobic conditions where biodegradation is slower (Chapter 3), although shallowly buried oil may be re-suspended by wave action or by ebullition of gases from the sediments. If the OPAs remain suspended, the additional reactive surface area afforded by the minerals or organic matter may enhance bacterial attachment (**Figure 2.3**) and aerobic biodegradation (Chapter 6).

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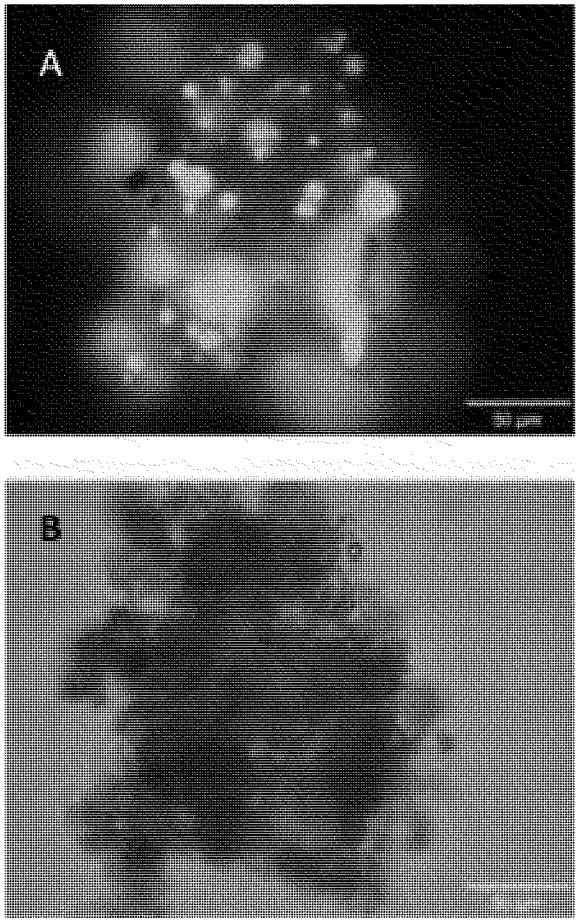

Figure 2.9 Oil-particle aggregate (OPA) formed by interaction of weathered dilbit with natural Kalamazoo River sediments. *A, appearance of an OPA under ultraviolet light; yellow areas are oil. B, the same OPA observed under visible light, showing oil plus sediment particles. Scale bars in bottom right are 50 µm. Images from Lee et al. (2012).*

OPAs can be generated by overland spills where oil initially contacts sediments on shorelines or riverbanks before entering the water column and sinking. Alternatively, oil that strands on beaches, interacts with shoreline sediments , then re-enters the water column with accumulated mineral particles may sink. Another natural mechanism for generating OPAs in the water is association of floating or submerged oil droplets with suspended particles in the water column, especially near the shore or in estuaries where the suspended sediment load is high. In fact, incorporating as little as 2 –3% mineral content into the oil phase is sufficient to cause some oils to sink (NRC 1999). The degree of interaction between oil and minerals depends on oil viscosity and the mineral type, but may begin with f ormation of an oil monolayer around the mineral followed by further accumulation of oil on the hydrocarbon surface (Omotoso et al. 2002).

The importance of submergence, OMA or OPA formation and sedimentation in the recovery and overall fate of spilled oil was highlighted in the DWH subsurface blowout in which chemical dispersants were injected into the escaping plume of light Macondo oil (**Table 2.2**). The large initial subsurface plume of suspended dispersed oil droplets (<100 µm diameter) at depths of 900-1,200 m was undetectable a few months after the discharge and was initially presumed by some researchers to have been rapidly biodegraded by indigenous bacteria (e.g., Hazen et al. 2010) that had been enriched by natural oil seeps in the Gulf (**Figure 2.3**). However, two independent analytical methods (hopane biomarkers and natural abundance radioisotopes) applied to sediment cores collected from the vicinity of the blowout have recently revealed elevated concentrations of hydrocarbons patchily distributed in a thin sediment layer overlaid by cleaner sediments from

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post-spill deposition (Valentine et al. 2014; Chanton et al. 2015; Joye 2015). This sequestered oil within a 3,200 km ^{2} footprint on the Gulf seabed may account for 2 -14% of the total volume of discharged Macondo oil (Valentine et al. 2014; Chanton et al. 2015; Joye 2015).

Two complementary mechanisms are proposed to have promoted this sedimentation pattern (Thibodeaux et al. 2011). One is the formation of OMAs as the circulating subsurface oil plume interacted with suspended mineral particles near the continental shelf, the concentrations of which may have further increased when flow from the Mississippi River was diverted to flush Macondo oil out of shorelines (Passow 2014). Sinking of these OMAs to t he seafloor (**Figure 2.3**) is proposed to have led to a 'bathtub ring' of oily sediments around the Gulf's continental shelf (Valentine et al. 2014).

A second mechanism of oil submergence, sinking and sedimentation, presumed to be biologically-driven, was al so observed by analysis of Gulf sediments and in laboratory simulations. In fact, this phenomenon may account for a large proportion of oil 'missing' from initial oil budgets for the DWH spill (e.g., FISG 2010; Joye 2015) and may be more important than OMA sedimentation in this particular spill. Creation of an oily 'footprint' or 'fallout plume', predominantly of microbial origin (Passow et al. 2012; Passow 2014; Valentine et al. 2014) may have formed as follows: even in uncontaminated ocean waters, marine phytoplankton and zooplankton near the sea surface naturally secrete sticky polymers that aggregate particulate and dissolved organic material into >0.5 mm flocs called 'marine snow' (Azam and Malfatti 2007 ; Kleindienst et al. 2015). These flocs slowly settle to the seafloor over months or years to form new sediments enriched in recalcitrant organic matter. During and briefly after the DWH event, unusually elevated concentrations and sizes of such flocs (mm to cm diameter) were observed (Brooks et al. 2015). These are proposed to represent blooms of hydrocarbon -degrading bacterial communities attached to weathered Macondo oil droplets at or near the sea surface (**Figure 2.3**). Additional organic material in the flocs may be mucus produced by oil -stressed phytoplankton. These large, dense , oil-rich organic flocs rapidly sank over days or weeks while incorporating additional submerged oil in the subsurface plume (and possibly mineral particles), creating a transient 'dirty blizzard' of oil-laden marine snow (Passow 2014; Valentine et al. 2014; Brooks et al. 2015). It is possible that some degree of aerobic oil biodegradation occurred during sinking of the OPAs, but the fallout area created an 'oily footprint' on the Gulf seafl oor. Thus, both surface and subsurface oil are thought to have been transported to the seabed in microbe-rich aggregates.

Weathering of Macondo oil at the surface, including photooxidation, affected floc formation (versus fresh Macondo oil that did not promote floc formation) and ma y have enhanced the rate of sinking of the oily marine snow (Chanton et al. 2014; Passow 2014). The role of Corexit 9500 dispersant in floc formation appears to be concentration-dependent (Passow 2014). Furthermore dispersion of Macondo-like oil with Corex it 9500 may lead to enrichment of specific *Colwellia* subtypes capable of growing on the dispersant itself and indirectly influencing floc formation (Kleindienst et al. 2015). These authors have recently claimed, based on laboratory experiments, that Corexi t 9500 may suppress oil biodegradation by indigenous hydrocarbon -degrading microbes in deep Gulf waters , although divergent opinions have been expressed regarding the effects of chemical dispersants on oil biodegradation (e.g., Prince 2015).

These types of hydrocarbon-rich sediment layers were gradually overlaid by clean sediments and post-spill marine snow through natural deposition processes, sequestering the oiled sediment layer (Romero et al. 2015). The long-term fate of this buried light Macondo oil is not yet known. However, it is possible that the sequestered oil will be biodegraded (even under anaerobic conditions in the sediment layer) because it is a light oil rich in alkanes, and because dissolution during initial rise of droplets to the surface (Section 2.4.2.1) and subsequent surface weathering (evaporation, photolysis) may have reduced the oil's toxicity. Also, attachment of active hydrocarbon-degrading microbes during floc formation may have 'inoculated' OPAs to promote

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biodegradation during and after sedimentation; however, this possibility remains to be investigated.

Even the relatively simple, well-characterized Macondo light crude exhibited unexpected sinking behaviour when dispersed. The submergence and sedimentation of chemically complex heavy oils and diluted bitumen blends is even more unpredictable, especially since dispersion of both fresh and weathered dilbits is not yet understood. Furthermore, there are very few published reports on formation of OPAs from heavy oil or bitumen products, especially in fresh water. This hampers understanding and modeling of dilbit submergence, incorporation into sediments, re-emergence and transport in rivers. The role of microbial flocs in sedimenting heavy residues into shallow receptors (e.g., river sediments and lakebeds) is unknown, as this mechanism was not recognized until very recently from DWH studies.

Formation of OPAs likely contributed to sedimentation of a diluted Bunker C-class heavy fuel oil spilled into Lake Wabamun, AB, by a train derailment in 2005 (Chapte rs 4 and 8). During the overland flow of this heavy oil into the lake the lighter diluent components evaporated, allowing the weathered oil to interact with shoreline material and subsequently with suspended organic particles in the water column. In additi on to forming tar balls (Section 2.4.2.4), some oil sank, impacted the lake sediments, and continued to resurface over subsequent years (Hollebone et al. 2011). Floating, sinking, submerged, sedimented and refloating oil, including the production of sheens, were all observed in the year following the spill (Hollebone et al. 2011).

Currently there is considerable debate regarding the fate of weathered and unweathered diluted heavy oils and bitumen blends : will they float o n the surface, submerge , sink or sediment with OPAs? An early literature review (NRC 1999) found that only 20% of heavy oil spills resulted in a significant portion of the products sinking or being submerged in the water. However, a more recent review (Winter and Haddad 2014) concluded that dilbit has a greater potential for sinking than conventional oils. First, evaporation of the diluent could increase the oil density, causing the residual oil to submerge or sink. Second, in theory , the majority of weathered dilbit would be available for interaction with suspended sediments as OMAs because so little of the bitumen itself will evaporate or biodegrade, further increasing the possibility of sedimentation and sequestration. It is not yet fully known how adhesive properties (which change with evaporative losses; Figures 2.5 and 2.6) affect OPA formation by diluted bitumen products in the field.

Field experience with the highly-publicized overland spill of dilbit into the Kalamazoo River, MI, in 2010 (Chapters 6 and 8) indicated that the dilbit, with an initial density of 0.96 -0.98 g/cm^3 and near neutral buoyancy, initially floated. Interaction with soil particles before entering the river and subsequently with elevated suspended sediment lo ads in the water due to high rainfall events preceding the spill enabled a considerable proportion of the dilbit to become submerged in the river water and eventually sink (Crosby et al. 2013). This phenomenon contaminated the river sediments at natural co llection points. The oil -sediment aggregates were stable for at least two years (Lee et al. 2012) , and at three years post -spill were estimated to represent 20 -30% of the initial oil volume (US-EPA 2013b).

This case supports the statement by Crosby et al. (2013) in a report from the US National Oceanographic and Atmospheric Administration (NOAA) that: 'Little research is currently available regarding the behavior of oil sands products spilled into water, and how they weather in the environment. Most tests h ave been conducted in the laboratory, so predicting the actual behavior of oil sands products for a range of spills is difficult.'

<u>Recommendation:</u> Additional study is needed to examine OPA formation by dilbits and heavy oil samples that have experienced va rious stages of weathering (including

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photooxidation, which may be important for heavy products that are enriched in PAH), using various types and loads of suspended sediments at various temperatures and salinities.

Some research is beginning to emerge to address this gap. Waterman and Garcia (2015) performed laboratory studies with CLB dilbit (**Table 2.2**) and Kalamazoo River sediments to examine the kinetics of OMA formation. The dilbit was artificially weathered (without UV exposure) over six days to achieve different mass losses (0%, 9.9% or 17.4% loss; density range 0.932–0.993 g/ml), then added to the surface of tap water containing uncontaminated Kalamazoo River sediments of different mesh sizes. With conventional gyrotory mixing at 21-22 °C the dilbit formed stable small-diameter (<200 μm) spherical droplets coated with particles (similar to those shown in **Figure 2.9** and in Stoffyn-Egli and Lee [2002]). However, larger (>1 mm) irregularly-shaped 'solid' OPAs formed with unconventional mixing intended to simulate sinking, re-suspension and droplet coalescence, possibly from folding of the sediments into the oil rather than simple attachment to the surface of droplets per the mechanism proposed by Stoffyn-Egli and Lee (2002). Some OMAs of each type were positively buoyant (floated in tap water), whereas others sank, depending on the relative proportions of oil and incorporated sediment. Thus, the behaviour of diluted bitumen is not predictable without experimental data to inform predictions.

Other recent laboratory studies in seawater found that weathered dilbit achieved densities >1 g/cm^3 and furthermore adhered to fine sediments (kaolin clay) added to the water, forming OMAs that sank in saltwater, as well as forming floating tar balls (Government of Canada 2013; reviewed by Dew et al. 2015). Additional details about floating and sinking behaviours of AWB and CLB dilbits in saltwater can be found in a Government of Canada (2013) report. The authors noted that bitumen is predicted to sink in fresh water after extensive weathering, but the behaviour of dilbit during weathering in the marine environment is currently poorly delineated. The relevance of using laboratory conditions to simulate oil behaviour in the environment has been questioned, as some studies have failed to use UV irradiation (to cause photooxidation) and/or to consider microbial interactions or temperature effects on density and sinking (e.g., weathered bitumen may float at 15 °C but sink at 5 °C; Section 2.2.1). As reviewed by Short (2013), other studies used unrealistically thick dilbit slicks or inappropriately slow wind speeds. To begin to address the need for open-air studies, experiments were performed on two fresh (unweathered) dilbit samples (AWB and CLB; **Table 2.2**) in outdoor wave tanks containing filtered seawater at ~19 °C (King et al. 2014). In only six days the AWB dilbit weathered sufficiently for droplets to detach from the slick and sink in the upper 10 cm of brackish water. This sinking was not enhanced by OMA formation because the seawater had been filtered to remove particles >5 μm. The density of CLB dilbit increased more slowly over 13 days, likely because of its naturally higher concentration of alkyl PAHs, which are more resistant to weathering (King et al. 2014).

Despite increased interest and research activity, such as laboratory and field tests that are currently underway in government laboratories (the results of which are not yet available for review), the conditions under which dilbit will float, submerge or sediment constitute a knowledge gap that warrants further investigation, particularly for freshwater systems. For example, the magnitude of microbially-driven sedimentation is currently cryptic, with or without photooxidation, OPA formation and different combinations of other weathering processes. If this sedimentation process is as significant for dilbit spilled in shallow water and freshwater systems as it appears to have been in the light Macondo oil subsurface spill, it is a major process that should be incorporated into oil spill models. The bimodal evaporation kinetics of dilbit and the longer time needed for onset and action of microbial processes to affect sinking versus physical processes (**Figure 2.4**) suggest that experiments on unconventional oils should be conducted for much longer time periods than trials with conventional crude oils.

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When oil sinks and becomes sequestered in sediments, it is less likely to be quickly or extensively biodegraded (Section 2.4.2.5) because the sediment pore spac es are usually anaerobic below the interface and nutrient replenishment is hindered. The fate of sequestered OPAs is largely unknown and requires study over long time periods (years) to account for the slow rates of anaerobic biodegradation. In the case of sequestered weathered dilbit and heavy oil, it is possible that the oil becomes virtually chemically inert to benthic organisms, although it may still have physical impact.

Recommendation: The long -term biodegradation of sedimented and sequestered oil in anaerobic marine and freshwater sediments requires long -term study (i.e., over years to decades) to provide reliable information for modeling and oil spill budgets.

2.4.2.4 Formation of tar balls, tar patties and tar mats

Tar balls are typically discrete, roughly spherical conglomerations of oil <10 cm in diameter, tar patties >10 cm in diameter, and tar mats ≥ 1 m in diameter (Fingas 2015b) that may be partially or completely submerged (Warnock et al. 2015). Pelagic tar floats or is shallowly submerged (and can travel considerable distances with waves and currents, where it can become stranded on the shore), whereas benthic tar residu es are immobilized by sinking to the sea floor. The impact of marine tar residues on aquatic life is less than that of fresh oil in the water column, although tar balls may serve as a reservoir of PAHs that partition into surrounding water (Martin et al. 2014), and tar residues that strand on beaches may release sheens of nearly -fresh oil if physically disrupted.

> Weathered oil that has acquired a surface 'crust' through weathering, emulsification, photooxidation and/or sediment interaction may form spherical tar balls, flattened tar mats or patties (collectively, marine tar residues) that may float, submerge or sink, depending on the oil type and severity of weathering.

Marine tar residues can be produced from 'chocolate mousse' emulsions (Section 2.4.1.4) and, due to entrained water, may harbour microorganisms wi thin the structure, as well as at the surface. However, the small surface area:volume ratios of tar residues, as well as the presence of a crust, hampers biodegradation of the oil (Warnock et al. 2015). The crust may be enriched in asphaltenes and resins as these are the most polar compounds in oil and are prone to alignment at the oil:water interface; therefore, heavy oils are more likely to form tar balls and mats. If suspended, the balls and mats may intermittently release a sheen of fresh oil from breac hes in the crust (Hollebone et al. 2011). Small tar mats may persist for decades, as noted after the 1979 Ixtoc I blowout in the Gulf of Mexico and the *Prestige* oil tanker sinking off Spain in 2002 (reviewed by Warnock et al. 2015). In 2013 an enormous tar mat estimated at 18 tonnes was discovered in the Gulf of Mexico as a consequence of the DWH subsurface spill in 2010. It consisted of 15% oil and 85% sand, shells and water (Buskey 2013).

2.4.2.5 Biodegradation

> Biodegradation is a natural process in which living organisms and/or their enzymes break down organic material to produce simpler chemical compounds such as organic acids, alcohols and/or gases. Bioremediation is the application of biodegradation processes to help clean up a polluted site. Petroleum biodegradation in aquatic environments is primarily achieved by bacteria.

Oil spill b iodegradation processes and limitations are discussed in detail in Chapter 3 and bioremediation in Chapter 6. Briefly, oil susceptibility to biodegradation primarily depen ds on: chemical composition (ratio of biodegradable alkanes and aromatics to recalcitrant resins and asphaltenes); physical state (surface area available at the

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oil:water interface for microbial attachment or dissolution of light hydrocarbons and hence also dependence on spreading, dispersion and emulsification state of the oil); temperature (biochemical activity being slower at low temperatures); nutrient availability (particularly soluble nitrogen and sometimes phosphate, as discussed in Chapter 6); and available electron acceptors for redox reactions (aerobic or various anaerobic conditions). Regarding the latter, oil degradation is generally faster and more efficient under aerobic conditions, such as in a well-aerated water column or in the uppermost layer of sediment, but may also occur slowly in buried sediments under anaerobic conditions (Section 2.4.3.1). The range of susceptible hydrocarbons under anaerobic conditions appears to be smaller than with aerobic degradation (e.g., reviews by Foght 2008; Mbadinga et al. 2011) and the enzymes, pathways, end products and residual oil composition also differ.

Oil biodegradation also requires competent microbes. Hydrocarbon-degrading bacteria are ubiquitous, and although they may be present in a pristine environment in only small numbers, they can flourish after an oil spill because they have an advantage over microbes that cannot utilize hydrocarbons. Environments that are routinely exposed to oil spills or natural hydrocarbon seeps, such as in the Gulf of Mexico, usually have indigenous microbial communities that are enriched in hydrocarbon-degraders and thus can respond more quickly to an incursion of oil. However, the widespread synthesis of alkanes by marine cyanobacteria may help maintain a baseline level of competent hydrocarbon-degrading bacteria even in pristine waters (Lea-Smith et al. 2015).

Notably, because biodegradation and physical weathering processes, such as evaporation and dissolution, target many of the same compounds (light hydrocarbons) but at different rates, it is often important to discriminate between biological and non-biological changes to oil chemistry. This can be done by the judicious selection of reference samples and the use of biomarkers (e.g., Blenkinsopp et al. 1996; Wang et al. 1998). As with chemical weathering, biological weathering is greatly influenced by oil composition, and each oil has different susceptibilities to aerobic and anaerobic biodegradation under specific environmental conditions. For this reason, surveys of oil biodegradability are useful for predicting oil fates. An example is the Environment Canada report by Blenkinsopp et al. (1996) in which a suite of oils transported in Alaska was tested in a laboratory for relative susceptibility to biodegradation by a standardized microbial consortium under cold freshwater conditions, with extended incubation times.

<u>Recommendation:</u> To complement catalogues of chemical and physical properties of oils, a biodegradability survey should be conducted on conventional and unconventional oils commonly transported in Canada. The survey should consider aerobic biodegradation potential under fresh- and saltwater conditions at cold and moderate temperatures, and anaerobic biodegradation in water-saturated sediments. If conducted under optimum laboratory conditions with fresh and weathered oils, the results would delineate the maximum biodegradation expected under actual environmental conditions.

The paucity of research on biodegradation of heavy oils and diluted bitumen must be addressed. It is certain that the majority of the bitumen in the blends is highly recalcitrant to biodegradation (Section 2.1.1.3 and 2.1.1.4). Even if some of the constituent bitumen chemicals were altered, would that change be detectable using current analytical methods (Appendix A) that are inadequate for characterizing asphaltenes and resins? Theoretically, the diluent components of bitumen blends should be relatively biodegradable (and resolvable by chromatography), but few blends have been subjected to controlled biodegradation experiments either aerobically or anaerobically to ascertain what proportion of the diluent is degradable in fresh or weathered dilbit. This uncertainty arises from the question raised in Section 2.3.1.2: does all diluent evaporate from diluted bitumen under natural conditions, leaving only recalcitrant bitumen? If so, there will be very little light hydrocarbon mass to be biodegraded in weathered dilbit. If not, is any of the residual diluent 'bioavailable' to microbes? It is possible that diluent chemicals are

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intimately associated with the resins and asphaltenes and therefore not available for microbial attack. Additionally, the diluent may be protected from microbial attack because the high viscosity of the weathered bitumen affords only a small surface area for microbial access.

Recommendation: Biodegradation of different bitumen blends and diluted heavy oils should be tested under a range of laboratory incubation combinations and analyzed using complementary suites of techniques to determine whether and how much biodegradation of constituents can be achieved. Comparison to multiple control conditions will be essential for rigorous interpretation of the results.

2.4.3 Oil Interactions with Shorelines and Sediments

Oil can impact shorelines and sediments via different routes. Overland spills will first impact the soil or beach so that the oil may be partially weathered by the time it reaches the water, as well as having incorporated mineral and organic matter (as discussed in Section 2.4.2.3). Oil that is spilled onto water and is subsequently stranded on the shoreline or riparian zone (the interface between a river or stream and the land) or buried in sediments will have different fates, including interaction with vegetation and possible re-mobilization.

2.4.3.1 Sorption, penetration and sequestration

Oil is adhesive, particularly when weathered (Fingas 2015b; **Figure 2.6**) and may attach to suspended particles (Section 2.4.2.3) and surfaces. If OPAs sink and are buried by additional sediment, the oil may be protected from remobilization, and biodegradation rates will be limited in instances where the water in sediment pore spaces is not adequately replenished with oxygen or nutrients. That is, oil buried in sediments, such as found in mudflats or deep ocean, may persist virtually unchanged.

Oil that remains floating or submerged in the water column can reach the shore and adhere to sand, cobbles, bedrock and shells, as well as man-made structures like piers and jetties; the same behaviour might be expected in the riparian zone. Once onshore in the intertidal zone, a light oil may penetrate by gravity into beach sediments due to its low viscosity, where it may sorb to the minerals and/or be retained (sequestered) in small pools, with reduced exposure to further weathering. This was observed after EVOS at many cobble beaches in Alaska. Relatively fresh oil can still be found in patches below the surface cobble and gravel more than 25 years after the spill (Atlas and Hazen 2011). This persistence is due in part to poor access of oxygen and nutrients to the buried oil, limiting remobilization and biodegradation of the oil (discussed in Chapter 3), as well as limited photooxidation, emulsification and other weathering processes (Boufadel et al. 2010).

Viscous heavy oils or highly weathered oils are less likely to penetrate deep into intertidal sediments, but may be forced to depth by wave action on high-energy beaches. Weathered oil that is thrown above the tidal zone will continue to experience physical and chemical weathering and may form an 'asphalt pavement', as observed on Chedabucto Bay beaches, persisting decades after the 1970 *Arrow* spill in Nova Scotia (Advanced Technology and Continental Shelf Associates 1990), and after the Baffin Island Oil Spill field experiment in the Canadian Arctic (Owens et al. 1987).

Although it is generally assumed that spills of light oils, such as condensate, in the open ocean would cause little or no harm due to rapid losses by evaporation, coastal studies in Nova Scotia have shown that condensate entrained within intertidal sandy beach environments can persist for extended periods (Strain 1986; Lee and Levy 1989). The implication is that evaporation of the diluent portion of dilbit that is sequestered in beach sediment may be retarded if unweathered dilbit is buried.

> Recent field observations and laboratory tests suggest that 'armoured beaches' (those that have boulders and/or cobbles underlain by fine sediments) may retain viscous heavy oils and heavily weathered oils, with the oil penetrating only to the surface of the packed sediment but not further.

On low-energy beaches, cobble and boulder 'armour' (**Figure 2.3**) prevents the sediments from being mixed by wave action and therefore protects the sequestered oil beneath (Harper et al. 2015). This oil, retained in the upper fine sediments, would be alternately replenished with oxygen and nutrients by tidal action and might undergo limited evaporation and dissolution, but the low surface area:volume ratio could slow biodegradation rates.

The presence of aquatic 'biofilms' (layers of living or recently living plants, animals and microbes) on rocks and structures could potentially reduce the adherence and sequestration of oil, particularly if the living material is highly hydrated (e.g., wet algae). The prevalence of algae on exposed bedrock shorelines is a component of ongoing baseline surveys in Canada that may yield observations about biofilm protection of substrata from oiling (Laforest et al. 2015).

There is much interest in the fate and behaviour of oil that enters aquatic areas having extensive vegetation (e.g., wetlands, estuaries, riparian zones) and impacts the underlying sediments. In this case, rates of submerged oil biodegradation may be retarded by anaerobic conditions in saturated sediments or by diauxic growth conditions (i.e., utilization of elevated levels of readily-degraded organic detrital matter by microbes, in preference to recalcitrant or toxic hydrocarbons). Alternatively, oil biodegradation may be stimulated by microbial associations with plant roots and the rhizosphere (soil surrounding roots and root hairs) in the process of 'phytoremediation' (**Figure 2.3**; Chapter 6). The latter interactions could also potentially contribute to biodegradation of components of the stranded residual oil in the sediments. Toxicity of dilbit spills to aquatic vegetation represents a current knowledge gap that has only recently begun to be studied (Dew et al. 2015).

2.4.4 Perspective

Based on review of the literature and on interviews conducted by the Expert Panel, it is clear that there is currently insufficient information about the chemical composition and environmental behaviour of emerging petroleum types, including diluted bitumen products, synthetic crude oils and perhaps shale oils. The question was raised by the Panel and others whether unconventional oils, such as diluted bitumens, are sufficiently different from conventional oils to be relegated to distinct classes for regulatory or spill response purposes. Based on current analytical capabilities, heavy oils and bitumen appear to fall along a recognizable chemical continuum spanning condensates to bitumen. However, in the case of diluted bitumen blends it is their physical behaviour in the environment, such as bimodal weathering, that currently appears to distinguish them from conventional petroleum and to confound extrapolation of their properties from the two components: diluent and bitumen. This lack of knowledge stems from insufficient information from laboratory tests, wave tanks or field trials to use in comparing bitumen blends with conventional oil behaviour, particularly in fresh water, in cold conditions and over prolonged time in the environment. For example, the basic question of how much diluent will evaporate from a particular diluted bitumen under environmental conditions has only begun to be addressed in laboratory and field trials. Emulsification of diluted bitumens is beginning to be tested under a variety of conditions, but much of the work to date has focused on seawater emulsification, with little work on fresh water. Similarly, new dynamic models are needed for predicting dissolution and dispersion of diluted bitumen, and there is a paucity of controlled trials showing the effects of particulates on dilbit sinking and sedimentation behaviour, whether accumulated from soils in overland spills or from interaction with suspended solids and microbes in the water column. Some of these parameters are currently being tested in wave tank studies with natural seawater,

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but the data are not yet available for the Panel to review. The US National Academy of Sciences (NAS) has commissioned a report on dilbit to be released shortly after this Report; the information in the NAS document may begin to address some of these gaps.

Recommendation: Each oil spill is unique due to the convergence of individual oil compositions with specific suites of environmental parameters. Therefore, the environmental behaviour of unconventional oils, blended heavy oils, bitumens and diluted bitumens needs to be investigated under a wide range of relevant environmental and climatic conditions, including long-term, large-volume outdoor trials. Online availability of such data would help inform first responders, remediation personnel and modelers. To conduct such research requires sustained funding and rapid access to funds set aside for investigating 'spills of opportunity'.

2.4.5 Oil Interactions with Ice

Petroleum resource development in the Arctic raises the potential of oil spills in cold, pristine, ice-impacted waters (Lee et al. 2011a). After having been given conditional regulatory approval in May 2015 and spending $7 billion to begin oil exploration in the Chukchi Sea off Alaska (Davenport 2015), Royal Dutch Shell PLC halted exploration in October 2015 citing reasons of insufficient quantities of oil, low crude prices, 'high costs associated with the project, and the challenging and unpredictable federal regulatory environment in offshore Alaska' (Katakey and Zhu 2015). In response the US Department of Interior cancelled the 2016 and 2017 auctions of natural gas and offshore oil leases (Kaufman 2015). Even with this recent turn of events, the likelihood of increased shipping traffic through navigable areas of the Northwest Passage brings greater risk for spillage of fuel oils, at the least. Furthermore, the concept of exporting Alberta bitumen from Churchill, MB, via tanker ships (Welch 2014) highlights the need for oil spill planning, preparedness and response in Arctic waters. Therefore, a brief summary of oil interactions with ice is presented here as a special case affecting oil behaviour, in addition to site-specific information presented in Chapter 3. Complete discussion of oil behaviour in/on ice and description of the different types of ice on water are beyond the scope of this Report; the reader is directed to comprehensive reviews (Lee et al. 2011a; AOSRT 2014; NRC 2014) and a list of ice definitions (Fingas 2015a). In addition, the long-standing, well-recognized Arctic and Marine Oilspill Program (now the AMOP Technical Seminar on Environmental Contamination and Response; http://ec.gc.ca/amop/) organized and sponsored by Environment Canada has published a large body of peer-reviewed physical, chemical and biological research, although much of it is not easily accessible. The reader is also directed to papers arising from the 1980-1983 Baffin Island Oil Spill (BIOS) project, several of which are published in a special issue of the journal *Arctic* (for an overview see Sergy and Blackall 1987). The BIOS project focused on oil spill countermeasures for Arctic near-shore and shoreline oil spills, particularly the use of chemical dispersants.

> Considering that large-scale oil spills, such EVOS in Alaska and DWH in the Gulf of Mexico, held surprises even for professionals in oil spill modeling, clean-up and response, the potential for a 'Coldwater Horizon' scenario in the Arctic Ocean is a daunting prospect.

Certain weathering processes discussed in Sections 2.4.1–2.4.3 (e.g., spreading, evaporation, dispersion, emulsification, biodegradation) are slower or may be eliminated seasonally at high latitudes under ice (e.g., photooxidation, formation of tar balls), whereas others may be enhanced (e.g., sinking). Some of these effects of ice are due to the increased viscosity and density of oil at low temperatures, to the lack of sunlight to form oxidized crusts on tar balls, or to the lack of open water. Generally, oil evaporation on water varies with the degree of ice coverage, wave height and air temperature (Brandvik and Faksness 2009). As ice forms, oil can remain on the ice surface (**Figure 2.10**), pool beneath it (Payne et al. 1990) or migrate upward through brine

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channels (Petrich et al. 2013) and fissures in the ice to the floe surface. In spring, if oil under the ice is warmed by sunlight, the lower albedo of the oil versus overlying snow and ice may allow the oil to surface through the ice and float on melt pools. Heavy or emulsified oil may be too viscous to migrate upwards and may remain pooled at the water–ice interface until disturbed by ice movement or melting. The smaller oil surface area and low temperatures decrease dissolution and dispersion into the underlying water and reduce evaporation into the atmosphere, thus retaining toxic LMW hydrocarbons. Conversely, evaporation may be the major process affecting oil stranded on the ice surface or absorbed into snow (Lee et al. 2011a). Oil trapped in pack ice will tend to move with the ice as it is driven by wind and currents, but spreading of oil on top of ice and snow is retarded, creating thicker, localized slicks. Spreading of oil under ice is much slower than on the water surface and the extent is largely dictated by the roughness of the under-ice surface; the ice:water topography may slow oil spreading (reviewed by Lee et al. 2011a). Emulsification may be reduced as ice dampens wind and wave action and as the oil is confined in thicker slicks within smaller open water areas (Brandvik and Faksness 2009). However, heavy fuel oils may remain suspended in ice–water slush (AOSRT 2014), rapidly becoming encapsulated in layers of ice during freeze-up (Dickins 2011) and isolating the oil from the underlying water, further reducing weathering. Interaction of oil with suspended minerals and organic particulates to form OPAs (Section 2.4.2.3) may still occur provided sufficient mixing, as observed in a field trial in the St. Lawrence River (Lee et al. 2011b).

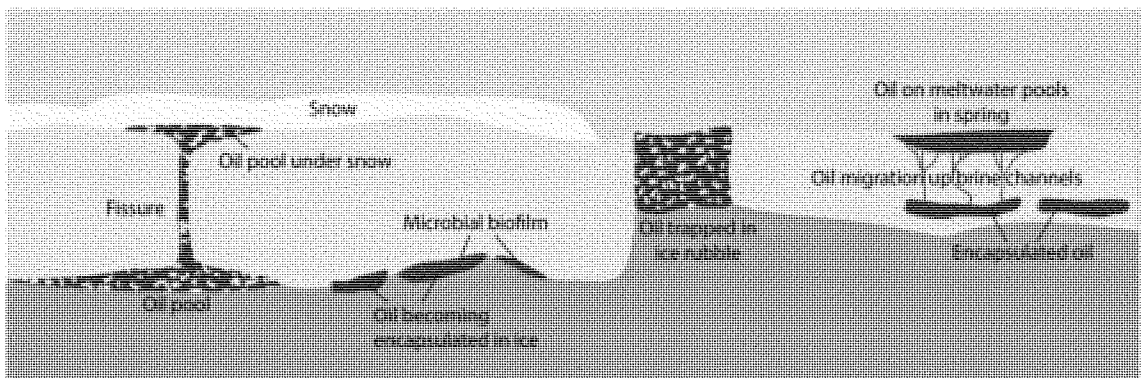

Figure 2.10 Summary of interactions of oil with ice and snow. *Additional detail can be found in AMAP (1998) Figure 10-5 and accompanying text.*

Cold marine waters are higher in dissolved oxygen and often also in dissolved nutrients, which should promote biodegradation, but the rates are likely slower in the Arctic than in more temperate water because enzyme processes and mass transport (discussed in Chapter 3) are slower at low temperatures. Nonetheless, biodegradation of two North Sea oils by microbes from Arctic Ocean water and sea ice has been demonstrated in the laboratory (Deppe et al. 2005), and versatile hydrocarbon-degrading bacteria have been recovered from seawater exposed to an experimental oil spill in the North Sea (Chronopoulou et al. 2015). Interestingly, diverse cold-adapted microbial communities, including species known to degrade hydrocarbons, can be found in biofilms up to several centimetres thick at the bottom of sea ice in contact with the water column along with pooled oil, where they may be able to biodegrade oil components (Greer et al. 2015). Recently, biodegradation of chemically- and physically-dispersed ANS oil was demonstrated at near-freezing temperature (-1 °C) by microbes in seawater from the Chukchi Sea, in the absence of additional nutrients (McFarlin et al. 2014). Arctic lakes, however, may present additional biodegradation constraints due to limited concentrations of nutrients and major ions.

The need for expanded testing of oil–ice interactions and oil behaviour in Arctic conditions merits special attention. Specifically, significantly more research is needed to understand the behaviour and fate of oil spilled into cold, deep seawater, including the rates and extents of physical, chemical and biological processes. Most published observations of oil–ice interactions have been made with first-year ice, but long-term studies of oil distribution and effects on multi-year ice are

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lacking. This is a time -sensitive knowledge gap, as increased shipping and the possibility of offshore drilling in Arctic waters could impact Canadian Arctic shorelines and seabeds. Likewise, future development of Canadian onshore Arctic resources and pipeline transport will demand investigation of oil effects on cold freshwater rivers and lakes and their porous shorelines.

Recommendation: Additional information is needed to describe and model the fates of petroleum in and under sea ice an d freshwater ice. Most of the limited research to date has been conducted on first-year sea ice, but studies with multi-year ice are needed.

The facilities to conduct such oil –sea ice interactions may soon be available at the newly announced Churchill Mari ne Observatory (Hudson Bay) to be constructed in Canada's only Arctic deepwater port (http://news.gc.ca/web/article-en.do?nid=996359). A purpose -built 'Oil in Sea Ice Mesocosm ' (OSIM; (http://news.umanitoba.ca/new-research-facility-to-open-in-churchill/) coupled with an Environmental Observing system will enable investigation, detection and mitigation of oil spills in Arctic sea ice.

2.5 Summary of Identified Research Needs and Recommendations

2.5.1 Research Needs

2.5.1.1 Short Term (High Priority)

Conduct research on the behaviour of unconventional crude oils, particularly diluted bitumen blends, spilled into marine and fresh water

The behaviour of unconventional oils and bitumen blends currently cannot be predicted with confidence. This knowledge gap affects risk assessments, spill response planning and cleanup decisions. Accelerated research is needed in the form of laboratory experiments and field trials to define the behaviour of diluted bitumen and unconventional crude oils (e.g., shale oils) under various seasonal and site conditions relevant to Canada.

Information needed about the effects of various weathering processes on such oils includes:

- Evaporation, particularly the bimodal behaviour of various bitumen blends;
- Emulsion formation, particularly of weathered diluted bitumen in fresh water;
- Susceptibility of dilbit spills to chemical dispersion;
- Sinking behaviour, including interactions with suspended particulates; and
- Susceptibility of fresh and weathered heavy oils and diluted bitumen products to biodegradation.

Combinations of environmental conditions, such as water temperature, salinity, mixing, aeration and sediment load, must be considered under laboratory and mesocosm (e.g., wave tank) test facilities over time scales of hours to days . Some trials are underwa y, but many relevant combinations of products and conditions exist and should be explored until general rules of oil behaviour can be discerned.

Information generated from such studies must be made readily accessible to first responders, regulators, remediation personnel, modelers and oil spill scientists.

2.5.1.2 Medium Term

Perform baseline physical, chemical and biological environmental surveys, including natural microbial communities and natural environmental variability.

To discern the impact of an oil spill we must have an understanding of the natural level of variability of sites under pre-spill conditions. Each oil spill is unique, not only to the specific oil

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product, but also to environmental and climate factors. It is thus essential to collect baseline physical, chemical and biological information for Canadian environments that are potential receptors of conventional or unconventional oil spills, particularly for sensitive ecosystems.

Of the factors reducing residual oil concentrations in the environment following a spill, biodegradation is the ultimate process for oil removal from the aquatic environment over the long-term. Therefore, the baseline surveys should include the composition of natural microbial communities in Canadian aquatic environments, particularly for sensitive ecosystems that may be impacted by spilled oil. Knowing the presence (or paucity) of natural hydrocarbon-degrading microbes in an ecosystem would help selection of cleanup strategies and predict the duration and success of cleanup efforts.

Information generated by the surveys must be made readily accessible to first responders, regulators, remediation personnel, modelers and spill scientists to support monitoring needs, the selection of response strategies and predictions of impact and recovery.

Develop a 'Biodegradation Index' that correlates with chemical composition of oils.

Twenty years ago Environment Canada was commissioned by the Alaska Department of Environmental Conservation to determine the susceptibility of nine crude oils and oil products commonly transported in Alaska to biodegradation under cold, freshwater conditions (Blenkinsopp et al. 1996). This type of study should be conducted using representative oils transported in Canada.

The study should include aerobic biodegradation potential under fresh- and saltwater conditions at cold and moderate temperatures, and anaerobic biodegradation in the presence of sediments. If conducted under optimum laboratory conditions, the results would delineate the maximum biodegradation to expect under actual environmental conditions. Furthermore, data generated from the screen should be used to construct a 'Biodegradation Index' correlating oil composition with susceptibility to biodegradation as a reference tool for oil spill remediation.

Augment online catalogues of crude oil composition and properties

Existing online catalogues, such as Environment Canada's Oil Properties Database and *CrudeMonitor.ca*, are incomplete in scope and in compositional information. Expanding and augmenting such catalogues would be useful for oil spill response planning and preparedness, particularly for diluted bitumen blends, and for the safety of first responders and the public.

2.5.1.3 Long Term

Support for fundamental petroleum chemistry research.

Fundamental understanding of oil composition and properties is lacking in several areas. These knowledge gaps can be addressed primarily in laboratory studies. Needs include:

- Development of validated and standardized methods to enable detailed chemical characterization of poorly-characterized oil components, including alkyl PAH, asphaltenes, resins, naphthenic acids and unresolved complex mixtures. Although this information may not have direct practical importance for oil transportation *per se*, it would provide insight into the fate and behaviour of spilled oil, including emulsification, photooxidation, biodegradation, toxicity, etc. This is a particularly critical gap for heavy oils and bitumen blends that have large proportions of these chemical classes, where such information should be used to refine oil spill models;
- Development of validated and standardized analytical methods and a framework to enable comparison of conventional and unconventional crude oils. This would help

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determine whether all transported oils simply reflect a continuum of chemical composition, or whether diluted bitumens represent a novel oil class. A forthcoming report by the US NAS may address part of the latter question, but the need for robust analytical methods remains; and

- Development of refined and standardized methods for measuring physical weathering processes, including evaporation, dissolution and oil:particle interac tions. Such information should be used to improve predictive spill models and inform mass balance budgets post-spill.

Apply research results to address the longstanding remediation question 'How clean is clean?'

Data obtained from laboratory studies, large-scale open-air experiments, long-term field research and baseline data surveys should be compiled and interpreted with the intention of determining acceptable endpoints of spill remediation based on chemical and biological indices. Long-term monitoring of follow-on effects after experimental spills and spills of opportunity would inform regulatory guidelines for cleanup endpoints.

2.5.2 Operational Preparedness Needs

Arctic preparedness: The re is urgent need for expanded documentation of oil–ice interactions and oil behaviour in Arctic conditions.

This is a time-sensitive knowledge gap, given predictions of commercial tanker traffic through the Northwest Passage and the potential for Arctic Ocean oil exploration a nd production. The facilities to conduct such oil-sea ice interactions may soon be available at the newly announced Churchill Marine Observatory in Hudson Bay. It is also important to note that most published observations of oil–ice interactions have been made with first-year ice, but long-term studies of oil distribution and effects on multi-year ice are lacking.

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CHAPTER 3: EFFECT OF ENVIRONMENT ON THE FATE AND BEHAVIOUR OF OIL

Abstract

Following Chapter 2 , which describes oil properties in terms of its chemistry and composition, this chapter is divided into three main topical areas, namely: 1) key environmental factors affecting spilled oil fate and behaviour , 2) oil impacts on different aquatic environments , and 3) importance of oil type in relation to the environment . Under each main section, up to nine subsections discuss in detail how th e biological, chemical and physical characteristics of each environment bring about change to the oil and the ramifications these changes have on the ecology of the biosphere. Section 3.1 outlines how microorganisms degrade the different hydrocarbon classes in oil and how temperature, dissolved oxygen, nutrient supply, salinity and pH affect and change the behaviour of the contaminating oil. Section 3.2 describes how oil impacts a wide diversity of environmental systems, such as freshwater ponds, lakes and rivers; estuaries, including bordering wetlands; seawater, both surface and deep sea; and Arctic environs, including permafrost, ice and snow. Section 3.3 concerns the three main types of oil (light, medium and heavy) and how they differ in their behaviour in relation to the various environments.

Research recommendations based on the discussion of knowledge gaps were consolidated and prioritized at the end of the chapter. In summary, research is needed to further our understanding of how the Arctic environment affects oil spilled into that harsh but sensitive ecosystem and how the spilled oil affects the biology and ecology of that environment. Oil interactions with permafrost and sprin g melt are important unknowns in terms of access to sites for spill response, rapid spreading of oil during freshet, slow rates of weathering, and how hydrocarbons interact with ice and suspended particulate matter. Research is also needed to further our understanding of the effects of spilled oil on permafrost ecosystems and how best to develop appropriate response strategies that mitigate the damage without causing further harm. In addition, little has been done to advance our knowledge of how to deal wit h subsurface blowouts in ice-covered marine environments. Given that Arctic drilling will likely ramp up in the future, improved methods will be needed to detect and monitor the behaviour of spills on, in and under ice and within the water column in the ev ent of such a subsurface deep sea blowout. While the National Energy Board has identified the need for additional precautionary measures (e.g., same season relief well capacity), further research is needed to aid in development of oil detection and respons e strategies in these important environments. Finally, although microorganisms represent the ultimate cleanup of oil -impacted environments, microbial activity is slower in temperatures near freezing. Yet such activity was found to be relatively fast in the 5 °C deep sea of the Gulf of Mexico during the Deepwater Horizon spill (Hazen et al. 2010). Confirmation of this finding would greatly advance our knowledge of the role microorganisms might play in Arctic spill cleanup in the future.

Introduction

Whereas Chapter 2 described the major physical and chemical properties that influence oil behaviour, Chapter 3 discusses what happens to oil spilled in various aquatic environments. Types of waterbodies that can be affected by oil spills include rivers, streams, m ud flats, wetlands (including bogs, fens and marshes), salt marshes, offshore (surface, water column and deep ocean), reservoirs, permafrost, muskeg, ponds, lakes and groundwater.

With or without human intervention, spilled oil is affected by microbes natu rally present in the environment that are in turn influenced by prevailing physical and chemical conditions at the spill site. Therefore, this chapter provides detailed discussion of key environmental factors that affect petroleum biodegradation, including indigenous microbial species, temperature, dissolved oxygen concentration, presence of nutrients, salinity (marine, estuarine and freshwater environments), pH and the type of environment impacted, such as shoreline type (e.g., lacustrine, riverine, marine and estuarine) and location (e.g., intertidal, supratidal, subtidal). Other factors include non -biological weathering processes,

oil type, amount spilled and components in oil causing short-term and long-term impacts. As discussed in Chapter 2, each type of oil has distinct physical and chemical properties that will influence the way it will spread and break down, the hazard it may pose to aquatic and human life, and the likelihood that it will pose a threat to natural and man-made resources. All these factors are discussed below in the context of aquatic environments.

3.1 Key Environmental Factors Affecting Spilled Oil Fate and Behaviour

Petroleum and its precursor materials have been present in the Earth's biosphere for tens of millions of years, and certain microbes have adapted to utilizing such high-energy compounds for growth. Even in the present time, some algae synthesize alkanes (Lea-Smith et al. 2015), calanoid copepods synthesize pristane from ingested chlorophyll (Short and Harris 1997), many plants synthesize waxes (Baker 1982), and petroleum naturally seeps from the earth's crust (such as in the Gulf of Mexico seabed, the Santa Barbara, CA, seafloor, at the sea bottom off the coasts of Baffin Island, NU, and the Queen Charlotte Islands, BC, and in the Athabasca River, AB, as it cuts through oil sands deposits). Thus, oil-degrading microbes are ubiquitous, and diverse microbes having the ability to degrade hydrocarbons will always occur in aquatic ecosystems, although in pristine sites they comprise a small proportion of the total microbial community. Oil incursion provides a competitive advantage for oil-degrading microbes compared to those that cannot utilize hydrocarbons. Oil degraders will thrive while the hydrocarbons are present and their proportions in a contaminated site will be much greater than in a pristine site, while overall microbial biodiversity decreases during active degradation. As the degradable components are depleted, the oil degraders will decrease in numbers and diversity will again increase.

> Hydrocarbons have been ubiquitous on this planet for millennia. Over this time, diverse microbes have adapted to consuming hydrocarbons under myriad different environmental conditions, typically cooperating as microbial communities to degrade complex oils.

Natural attenuation is the result of petroleum biodegradation by oil-degrading microbes naturally present in the biosphere and is unassisted by humans. These natural rates of oil degradation may be enhanced by bioremediation, which is a human intervention practice that accelerates the rate of oil decomposition by microbes either through biostimulation (a process used to accelerate biodegradation by supplying the indigenous oil-degrading communities with nutrients or other growth needs, as discussed below, or bioaugmentation (addition of exogenous oil-degrading bacteria to augment the natural populations to degrade the contaminating hydrocarbons). Bioremediation is normally used as a polishing step after oil that can be removed by other means has first been removed, so that the microbes have only a small mass of oil to degrade. These processes are discussed in greater detail in Chapter 6. Regardless of intervention, microbial hydrocarbon degradation is still subject to physical and chemical conditions in the impacted environment.

3.1.1 Presence of microbial species in oil-impacted water, sediments, shorelines and wetlands

Much progress has been made in recent years in our knowledge of hydrocarbon-degrading microbial communities, especially in marine ecosystems. This research has identified a group of aerobic microorganisms called obligate hydrocarbonoclastic (hydrocarbon-degrading) bacteria (OHCB), which have been shown to play a key role in eliminating petroleum hydrocarbons contaminating marine waters (natural seeps as well as accidental spills). According to Yakimov et al. (2007), when hydrocarbon contamination occurs in marine ecosystems, successive blooms occur in indigenous bacterial genera belonging to the OHCB group (*Alcanivorax*, *Cycloclasticus*, *Marinobacter*, *Thallassolituus* and *Oleispira,* among others), which were present at undetectable levels prior to the spill. The most notable genera are *Alcanivorax,* which contains several key enzymes that confer the ability to metabolize alkanes for carbon and energy acquisition (Schneiker et al. 2006), and *Cycloclasticus,* which degrades polycyclic

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aromatic hydrocarbons (PAHs) (Table 2.1). Microbes, such as *Alcanivorax borkumensis,* are exquisitely adapted to biofilm for mation (Figure 2.3), oil solubilization, degradation of *iso*-alkanes (Table 2.1) that are considered recalcitrant, and tolerance of oil -induced stress (Schneiker et al. 2006; Sabirova et al. 2008). In addition to OHCB, many generalist (non -obligate) hydrocarbon -degraders, such as *Pseudoalteromonas,* are also prevalent in marine water (e.g., in the North Sea; Chronopoulou et al. 2015). OHCB have not yet been reported in marine sediments, shorelines and wetlands, although these environments do harbour diverse generalist oil-degrading microbes. The difference in sediments and soils may be due to anaerobic conditions versus aerated/aerobic conditions in surface water (Section 3.1.3), resulting in the presence of different predominant microbes.

Some bacterial species are specialists in degrading saturated hydrocarbons and others in aromatics (Foght et al. 1990). Thus, a suite of different microbes (a 'community') is required to achieve maximum oil biodegradation. Some microbes possess complete biodegradation pathways and therefore can mineralize hydrocarbons, oxidizing them completely to carbon dioxide (CO_2) and water without requiring a partner microbe. Mineralization is the preferred outcome for oil spill cleanup. However, some microbial species have incomplete pathways and are only able to partially oxidize ('transform') the hydrocarbon, especially if it is complex (e.g., an alkyl PAH). In the latter case, the transformation product from the first attack may subsequently be mineralized by other species in a combined or concerted effort by a microbial community, or it may resist further degradation and accumulate. These partially oxidized products may be more or less toxic than the original substrate, depending on the chemical properties of the transformed product. Furthermore, some bacteria have the ability to exchange gene tic material (DNA) so that species previously unable to degrade hydrocarbons may acquire that ability by obtaining some or all of the genes encoding the requisite enzymes. Thus, the products of microbial oil biodegradation can vary depending on the composition of the oil, the microbial community and gene transfer among species.

The enrichment of the aforementioned bacterial genera is affected by temperature, salinity, oxidation - reduction potential and other physical/chemical factors. *The rapid development o f these classes of microorganisms can be accelerated by the use of biostimulation procedures , such as the addition of nutrients (for example, nitrogen and phosphorus) that frequently exist only in limiting quantities in nature (discussed in greater detail in Chapter 6).*

Many of the constituents of petroleum oils have been known to be biodegradable for decades. Biodegradation of oil is one of the most important processes involved in weathering and the eventual removal of petroleum from the environment, parti cularly for the non -volatile components of petroleum. Numerous scientific review articles have covered various aspects of this process and the environmental factors that influence the rate of biodegradation (Zobell 1946, 1973; Atlas 1981, 1984; N RC 1985; Leahy and Colwell 1990; Johnsen et al. 2005; Head et al. 2006; Yakimov et al. 2007; Singh et al. 2012; Gutierrez et al. 2013). Two comprehensive EPA reports on oil spill bioremediation covering all aspects of microbial interactions with spilled oil and prov iding guidelines on how to implement bioremediation in the field were published in 2001 and 2004 and are still relevant today (Zhu et al. 2001, 2004).

Microorganisms capable of degrading petroleum hydrocarbons and related compounds are ubiquitous in marine, freshwater and soil habitats (Head et al. 2006; Yakimov et al. 2007; Atlas and Hazen 2011), and degradation mediated by indigenous microbial communit ies is the ultimate fate of most of the hydrocarbons that contaminate the environment (Leahy and Colwell 1990; Foght 2008; Prince 2010; Atlas and Hazen 2011; Kostka et al. 2014). Hundreds of species of bacteria, yeasts and fungi have been shown to degrade hydrocarbons ranging from methane to compounds of over 40 carbon atoms and in temperatures spanning psychrophilic (-20 °C to +10 °C) to thermophilic (41°C – 122 °C) (Zobell 1973; Floodgate 1984; Jordan and Payne 1980; Cooney 1984; Margesin et al. 2013).

Despite their ubiquity in nature, hydrocarbon degrading microorganisms are distributed in relation to the historical exposure of the environment to hydrocarbons. Those environments with recent or chronic oil

contamination, in particular the Gulf of Mexico with its myriad natural petroleum seeps, will have a higher percentage of hydrocarbon degraders than unpolluted areas. Notably, no single strain of bacteria has the metabolic capability to degrade all the components found in crude oil (Karrick 1977; Leahy and Colwell 1990; Vinas et al. 2002). In nature, biodegradation of a crude oil typically involves a succession of species within the consortia of microbes present. Microorganisms classified as non-hydrocarbon utilizers may also play an important role in the eventual removal of petroleum from the environment. Degradation of petroleum involves progressive or sequential reactions, in which certain organisms may carry out the initial attack on the petroleum constituent, producing intermediate compounds that are subsequently utilized by a different group of organisms, in the process that results in further degradation.

Petroleum is a complex mixture of many thousands of compounds consisting mostly of carbon and hydrogen. As mentioned in Chapter 2, these compounds have been divided into four major groups, abbreviated as SARA (saturates, aromatics, resins and asphaltenes). In general, the saturates fraction is the most biodegradable with the exception of the fused complex alicyclic ring compounds, such as the hopanes and steranes. The aromatic compounds, especially the PAHs, are of intermediate biodegradability (which is inversely proportional to the number of fused rings), whereas the polar fractions (i.e., resins and asphaltenes) have been long thought resistant to biological degradation (Pineda-Flores et al. 2004; Naveenkumar et al. 2010).

With respect to saturates, the predominant mechanism of *n*-alkane degradation involves terminal oxidation to the corresponding alcohol, aldehydes or fatty acid functional group. Branched alkanes are less readily degraded in comparison to *n*-alkanes. Methyl branching increases the resistance to microbial attack because fewer alkane degraders can overcome the blockage of beta-oxidation (NRC 1985). Highly branched alkanes, such as pristane and phytane, which were earlier thought to be resistant to biodegradation, have also been shown to be readily biodegradable, although at lower rates than normal alkanes. Cycloalkanes, however, are more resistant to biodegradation. Complex alicyclic compounds, such as hopanes and steranes (see Table 2.1), which belong to the class of pentacyclic triterpanes, are among the most persistent compounds of petroleum spills in the environment (Wenger et al. 2002; Peters et al. 2005). Because of their relative resistance to microbial attack, these compounds are used to normalize the concentrations of the more biodegradable petroleum compounds as a chemical forensic tool.

By normalizing biodegradable petroleum hydrocarbons to the relatively non-biodegradable hopane, the variability of loss among alkane and aromatic compounds can be significantly mitigated, thereby facilitating the calculation of rates of biodegradation of petroleum hydrocarbons in field studies.

Although the aromatics are generally more resistant to biodegradation than the alkanes, some low molecular weight (LMW) aromatics, such as naphthalene, may actually be oxidized before many saturates (Foght 2008). Monoaromatic hydrocarbons, such as benzene or toluene, are toxic to some microorganisms due to their solvent action on cell membranes, but in low concentrations they are easily biodegradable under aerobic conditions. PAHs with 2-4 rings are less acutely toxic and biodegradable at rates that decrease with the level of complexity. PAHs with five or more rings can only be degraded through cometabolism, in which microorganisms fortuitously transform non-growth substrates while metabolizing simpler hydrocarbons or other primary substrates in the oil. Alkylated aromatics are degraded less rapidly than their parent compounds; the more highly alkylated groups are degraded less rapidly than less alkylated ones. The metabolic pathways for the biodegradation of aromatic compounds have been the subject of extensive study (Atlas 1981; Cerniglia 1992; Prince 1993). The bacterial degradation of aromatics normally involves the formation of a diol (two adjacent hydroxyl groups attached to the aromatic ring), followed by ring cleavage and formation of a di-carboxylic acid. Fungi and other eukaryotes normally oxidize aromatics using monooxygenases, forming a trans-diol.

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Compared to saturates and aromatics, little is known about biodegradation of resins and asphaltenes due to their complex structures, making them difficult to analyze. Resins and asphaltenes have previously been considered to be refractory to biodegradation. However, recent evidence suggests that asphaltene degradation may occur through cometabolism (Leahy and Colwell 1990) or by other means (Liao et al. 2009).

Biodegradation typically affects the physical properties of oil by causing a decrease in the LMW alkane and aromatic fractions and enrichment in the polar resins and asphaltene fractions (Liao et al. 2009). The increase in the fraction of asphaltenes in heavy oil is attributed to the selective loss of alkanes and some aromatic hydrocarbons. The asphaltene fraction is defined typically to contain those species that are insoluble in low carbon number *n*-alkanes, such as *n*-hexane or *n*-heptane, while resins are soluble in such solvents (Tissot and Welte 1984). In their study of biodegradation of asphaltenes, Liao et al. (2009) concluded that the products of biodegradation, such as carboxylic acids, phenols and alcohols, may not only contribute to the resin fraction of crude oils, but are also linked with functionalities of resins and asphaltenes. The amount of asphaltenes increases because some resin molecules are enlarged and their polarity increased such that they can be precipitated by hexane as newly generated asphaltenes.

One study (Uraizee et al. 1998) involved systematically altering the concentration of non-biodegradable material in crude oil and analyzing its impact on transport of the biodegradable components of crude oil to the microorganisms. They did this by first fractionating crude oil into its primary SARA classes, then reconstituting the oil with increasing concentrations of the asphaltene fraction, and finally subjecting the reconstituted oil to biodegradation in the laboratory as measured by respirometry. The authors developed a mathematical model to explain and account for the dependence of biodegradation of crude oil through a putative bioavailability parameter. Experimental results indicated that as the asphaltene concentration in oil increased, the maximum oxygen uptake in respirometers decreased. The mathematically fitted bioavailability parameter of degradable components of oil also decreased as the asphaltene concentration increased. This was offered as one explanation for the lower biodegradation rates of heavy oils vis-à-vis lighter oils.

Mass transfer is often overlooked as an important aspect of hydrocarbon biodegradation. If one assumes that the aqueous solubility of PAHs limits their microbial degradation, mass transfer becomes important in understanding and predicting the fate and rate of biodegradation in aqueous solution (Volkering et al. 1992, 1998; Harms and Bosma 1997; Bosma et al. 1997; Johnsen et al. 2005). Optimization of mass transfer can be achieved in several ways:

- The production of biosurfactants by the degrading microbial communities, which facilitate transmembrane diffusion of hydrocarbons into the cell by increasing the diffusion coefficient governing the flux, and
- Exogenous addition of surfactants, such as dispersants, which tend to increase the interfacial area, also resulting in a higher flux of substrate (PAH).

The rate at which microorganisms degrade a substrate at low concentration depends on their specific affinity toward the substrate (Harms and Bosma 1997), defined as the ratio of the maximal substrate uptake rate to the half saturation constant (Button 1985). High specific affinities lead to efficient low-concentration PAH removal from the environment at the cell surface, steeper concentration gradients and higher substrate transfer rates.

In addition to affecting diffusion as discussed above, surfactants also influence dissolution and subsequent uptake of PAHs into the cell by attaching to the oil-water interface where micelles facilitate PAH release from the interface (Volkering et al. 1998). Alkane degraders are known to produce biosurfactants to increase the bioavailability of the poorly soluble hydrocarbons (Beal and Betts 2000; Noordman and Janssen 2002). Such biosurfactants, especially those produced by *Pheudomonal*

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Hydrocarbon biodegradation can occur over a wide range of temperatures, but rates generally increase with increasing temperature to an optimum for each environment.

aeruginosa, are commonly comprised of rhamnolipids, which are characterized by a sugar head group and a fatty acid tail. However, biosurfactant production is not as common among the PAH degraders (Itoh and Suzuki 1972; Oberbremer and Muller-Hurtig 1989; Johnsen et al. 2005), which means that biosurfactants may not be as important for PAH metabolism as for alkanes. In fact, because surfactants are known to interact directly with cell membranes, surfactants such as those in dispersants may negatively affect uptake of PAHs into the cell and subsequent metabolism by disrupting the integrity of the membrane, and it is possible this may even be exacerbated at deep sea hydrostatic pressures (Campo et al. 2013).

Studies involving hydrocarbon-degrading communities, mostly conducted in laboratory cultures, have been very helpful in elucidating the role of oil biodegradation in nature. However, laboratory studies can also be misleading to our complete understanding of the behavior and fate of oil when released into the environment. A fuller understanding of the dynamics of *in situ* biodegradation affecting our ability to predict the ultimate fate of hydrocarbons remains a challenge (Prosser et al. 2007).

3.1.2 Temperature

Temperature exerts major effects on the physical properties of oil, such as viscosity, surface tension and density, that may influence the oil's physical transport and environmental persistence, including biodegradability, within the environment (Chapter 2). Temperature also influences the activity and composition of the microbial community that responds to oil.

3.1.2.1 Physical-Chemical Properties of the Oil Influenced by Temperature

At low temperatures, the viscosity of the oil increases (Figure 2.2) and the volatility of toxic LMW hydrocarbons declines, delaying the onset of biodegradation (Atlas 1981). This property may be somewhat advantageous when a spill occurs in cold regions because it will allow response teams more time to deploy treating equipment. Some hydrocarbons are more soluble at lower temperatures (e.g., short-chain alkanes) and some LMW aromatics are more soluble at higher temperatures (Polak and Lu 1973), altering susceptibility to biodegradation. Specific gravity of oil is dynamic in that it will increase over time as the lighter components within the oil evaporate, dissolve or biodegrade, and it also responds to temperature. Heavier oils, such as refined oils, dilbit and Orimulsion ™ (bitumen mixed with water), may sink and form tar balls at low temperatures or may interact with rocks or sediments on the bottom of the water body.

3.1.2.2 Effect of Temperature on Biodegradation

Figure 3.1 shows that highest degradation rates generally occur in the range of 30–40 °C in soil environments, 20–30 °C in some freshwater environments, and 15–20 °C in marine environments (Jordan and Payne 1980; Bartha and Bossert 1984; Cooney 1984). These general increases with temperature reflect the reaction rates of enzymes required for oil biodegradation, which generally double with a temperature increase of 10 °C, to an optimum for each enzyme type. The variation among media reflects the preferred growth temperatures of the different microbial communities that have adapted to their environment. In addition to direct biochemical effects, increased temperatures can reduce a liquid's surface tension, making oil more likely to spread on warmer water than very cold water, thus affecting the surface area of oil available for interaction with microbes.

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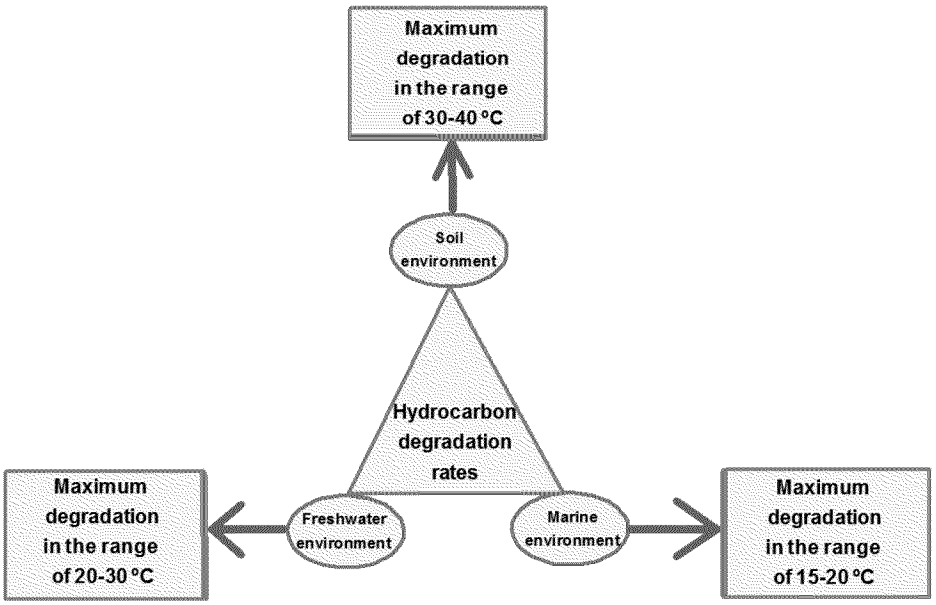

Figure 3.1 Optimum temperatures for hydrocarbon degradation rates in soil, fresh water and marine environments. *Adapted from Das and Chandran (2011).*

Although biodegradation typically occurs faster and more completely at higher temperatures, in environments where a cold-adapted microbial community has been established (e.g., one that prefers temperatures < 15 °C, such as in perennially cold environments), degradation can also occur at significant rates. Hydrocarbon biodegradation has been observed at temperatures as low as 0-2 °C in seawater and -1.1 °C in soil. Colwell et al. (1978) reported greater degradation of Metula crude oil at 3 °C than at 22 °C with a mixed culture in beach sand samples. Westlake et al. (1974) found that bacteria capable of degradation at 4 °C would metabolize oil at 30 °C, but those populations that developed at 30 °C had a limited activity at 4 °C. Lee et al. (2011) collected samples from the St. Lawrence River during a winter field trial study following the release of Heidrum crude oil (25 °API; Chapter 2) in ice-infested waters, and used the samples to monitor time-series changes in oil concentrations and composition in laboratory microcosms. Detailed chemical analysis (GC-MS with hopane as the normalizing biomarker; Appendix A) from these studies showed that more than 60% of the total petroleum hydrocarbons, 75–88% of total alkanes, and 55–65% of total PAHs were degraded after 56 days of incubation at 0.5 °C. McFarlin et al. (2014) recently reported on the biodegradation of Alaska North Slope crude oil in Arctic seawater collected from the Chukchi Sea, AK, incubated at −1 °C. Indigenous microorganisms degraded both fresh and weathered oils, in both the presence and absence of the chemical oil dispersant Corexit 9500, with oil losses ranging from 46−61% and up to 11% mineralization over 60 days.

> Temperature affects microbial communities and oil properties in complex ways, so the rate and extent of biodegradation does not have a simple relationship with temperature.

A review by McGenity et al. (2012) presented strong evidence for oil biodegradation in cold marine environments. Instead of *Alcanivorax*, which is a halophilic [1] oil degrader found mostly near the ocean's surface and able to grow at temperatures from 4 to 35 °C, the alkane-degrading obligate psychrophile [2] *Oleispira* has been commonly associated with biodegradation of oil spills in cold marine environments and in fact can constitute as much as 90% of the microbial community near the oil spill. This genus has a

[1] *Halophiles are species that require a substantial concentration of dissolved salt to survive.*
[2] *Psychrophiles are species that require constant low temperatures (usually < 10 °C) and die at higher temperatures.*

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wide global distribution and other genera are being isolated and characterized, such as *Oleibacter* sp. (Teramoto et al., 2011) and microorganisms belonging to the order *Oceanospirillales* (Golyshin et al. 2002; Hazen et al. 2010). Likewise, the bacterial genus *Thalassolituus* is a marine OHCB that has been observed as the dominant member of an oil-degrading microbial community in the Arctic seawater surface (Yakimov et al. 2007). Understanding presence, abundance, activity and needs of cold-adapted hydrocarbon degraders is becoming important with the announcements of offshore oil and gas exploration at high latitudes in regions such as the Chukchi and Beaufort Seas and with anticipated increases in shipping in the Northwest Passage (http://www.alaskapublic.org/2015/07/31/shells-exploratory-drilling-commences-in-the-chukchi-sea/). However, more recent research casts doubt on future utility of transportation in the Northwest Passage as findings by Haas and Howell (2015) show that even in today's warming climate, ice conditions must still be considered severe (i.e., thick ice more than 100 m wide and thicker than 4 m has been observed to occur frequently).

Recommendation: Confirmation of the finding that microbial activity was relatively fast in the 5 °C deep sea of the Gulf of Mexico during the Deepwater Horizon spill (Hazen et al. 2010) would greatly advance our knowledge of the role microorganisms might play in Arctic spill cleanup in the future.

3.1.3 Dissolved Oxygen

Various microbes are able to grow with or without oxygen for their metabolism, and some can alternate depending on the available oxygen in their environment. Oxygen has finite solubility in water (~7.5 mg/L at 30 °C, increasing to ~15 mg/L at 0 °C for water in equilibrium with air at sea level) and, if oxygen is depleted by microbial activity, it must be replenished to maintain aerobic metabolism. Replenishment is readily accomplished in most shallow water columns by wind and wave action, so conditions of oxygen limitation normally do not exist in the upper levels of the water column in marine and freshwater environments and in the surface layer of most beach environments. Therefore, dissolved oxygen is unlikely to limit biodegradation rates in these environments except under certain conditions, such as strong stratification, although the physical state of the oil and the amount of available substrates also affect oxygen availability.

Aerobic conditions are generally considered necessary for extensive degradation of oil hydrocarbons in the environment since major degradative pathways for both saturates and aromatics involve oxygenases (Atlas 1981; NRC 1985; Cerniglia 1992), and biodegradation of petroleum in the field has been shown to occur substantially faster under aerobic than anaerobic conditions. Although aerobic degradation of crude oil has been well-known for decades (e.g., Atlas 1981), mechanisms are still being elucidated for both alkane and PAH metabolism (Rojo 2009; Kanaly and Harayama 2010; Pérez-Pantoja et al. 2010; Fuchs et al. 2011).

> The major pathways for removal of hydrocarbons from the environment involve aerobic conditions.

There is little doubt about the importance of oxygen in determining the rates and pathways of hydrocarbon biotransformation in ecosystems that are dominated by anaerobic and/or anoxic conditions including: subsurface sediments (marine, riverine and lacustrine); most fine-grained marine shorelines; coastal and inland wetlands beneath the top several millimeters of soil; salt marshes; mud flats; groundwater; muskeg; bogs and fens; anoxic zones of water columns; and any others where oxygen is limiting. Bottom sediments are typically stratified, with dissolved oxygen and redox potential[3] decreasing sharply with depth, often to 0 mL L^{-1} and -300 mV or less, respectively—

[3] *In environmental chemistry, redox potential is an electrochemical measurement or calculation to determine whether reducing or oxidizing conditions prevail. Aerobic sites usually have positive redox values and anaerobic sites have progressively more negative redox values.*

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i.e., stringently anaerobic conditions. Anoxia is primarily a result of the high affinity of aerobic micro-organisms for oxygen, leading to its depletion in sediments where oxygen cannot diffuse quickly enough from bottom water into pore water to replenish concentrations and maintain aerobic conditions.

Under anaerobic conditions the ultimate degradation of organic matter is generally much slower (Head and Swannell 1999). Anaerobic degradation mechanisms involve different microbial species, degradation enzymes and pathways, metabolites and end products than aerobic biodegradation and apparently also impose a narrower substrate range (Schneiker et al. 2006; Sabirova et al. 2006; Foght 2008). Furthermore, whereas a single species can usually accomplish complete aerobic hydrocarbon mineralization using well-studied pathways, anaerobic hydrocarbon biodegradation has only recently been recognized, many of the key microbes and their pathways are still cryptic, some metabolites are unknown, and need for a community of microbes appears to be more prevalent. Anaerobic oil degradation was deemed in early studies to occur only at negligible rates, leading to the conclusion that the environmental importance of anaerobic hydrocarbon degradation could be discounted in terms of remediation response. More recently, the biodegradation of some aromatic hydrocarbons, such as BTEX compounds and a few PAHs (Foght 2008), *n*-alkanes (Mbadinga et al. 2011) and *iso*-alkanes (Abu Laban et al. 2014), has been clearly demonstrated to occur under a variety of anaerobic conditions. Studies have also demonstrated that in some marine sediments, PAHs and alkanes can be degraded under sulfate-reducing conditions at similar rates to those under aerobic conditions (Coates et al. 1997; Caldwell et al. 1998). *Regardless of the large gaps in understanding anaerobic hydrocarbon metabolism, the occurrence of anaerobic environments in marine ecosystems is sufficiently great that the anaerobic fate of crude oil must be considered and the overall importance of anaerobic biodegradation of crude oil in the environment requires further study.*

> Anaerobic hydrocarbon metabolism is emerging as an important mechanism for hydrocarbon biodegradation.

It is important to note that all biodegradation of crude oil, whether anaerobic or aerobic, freshwater or marine, aquatic or terrestrial, is selective in that some components are only partially degraded because of their size or complexity or not modified at all (Wang et al. 1998). Thus, *there will always be a residue of original petroleum components, like asphaltenes and resins, partly degraded intermediates and other end products of mineralization that may be generated (CO_2, CH_4, H_2S, etc.), the mass of which will depend on the interplay between environmental conditions, petroleum chemistry and microbial community composition.*

<u>Recommendation</u>: Further research is required on the mechanisms, pathways and rates of anaerobic biodegradation of hydrocarbons, particularly with respect to circumstances where oil has penetrated into sediment (riverine, lacustrine, estuarine or marine) where it could persist for decades.

3.1.4 Nutrient supply and stoichiometric ratios of C:N:P

The incursion of crude oil into an environment represents a large influx of carbon that is normally nutrient-poor. In particular, nitrogen (N) and phosphorus (P) are required for balanced growth of microbes that biodegrade the oil and other microbial community members that comprise the base of the food chain. In the 1930s, Alfred Redfield made thousands of measurements of the stoichiometric ratios of C:N:P in oceanic plankton (Redfield 1934). These ratios were strikingly similar to the dissolved concentrations of N and P found in the deep sea. This C:N:P relationship became known as the Redfield Ratio, which is 106:16:1, and has stood the test of time. Thus, in marine environments, nutrient limitation is generally correlated to the low background levels of N and P in seawater. The C:N:P stoichiometry of phytoplankton ultimately controls the nutrient ratios of the deep ocean (Arrigo 2005). These interactions produce a self-regulating, biogeochemical system that maintains quasi-stable oceanic nutrient inventories over both short and long timescales (Arrigo 2005). These stoichiometric ratios are important in

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understanding the role microbes play in the event of a catastrophic oil spill when carbon is in great excess to N and P. *Once the oil is beached, the only way to accelerate biodegradation is to add limiting nutrients in appropriate amounts to produce approximately the Redfield Ratio.*

Freshwater systems (lakes and wetlands) range from oligotrophic[4] to eutrophic[5]. Rivers may be nutrient-poor at the source but generally become nutrient-rich downstream after receiving industrial and domestic effluents and agricultural runoff (Cooney 1984). Freshwater wetlands are typically considered to be nutrient limited, due to heavy demand for nutrients by plants. They are also viewed as being nutrient traps, as a substantial amount of nutrients may be bound in biomass (Mitsch and Gosselink 1993). Both freshwater lakes and wetlands may also exhibit seasonal variations in nutrient levels, which will affect the performance of oil biodegradation. *Ward and Brock (1976) found that the rate of hydrocarbon biodegradation in an oil-contaminated lake was greatest during early spring when the nutrient content was also high. As N and P levels declined in the summer (probably due to uptake by algal growth), oil biodegradation also decreased.*

Thus, although oil is rich in energy, carbon and hydrogen needed for microbial growth, it is poor in essential nutrients, such as N and P. To balance the 'diet' of oil-degrading microbes, these nutrients can be added in water-soluble form (such as conventional fertilizers) often, but not always, in ratios of C:N:P approximating the 106:16:1 Redfield Ratio, such as the commonly used ratio of 100:10:1. Adding nutrients is particularly important for beached oil, which no longer has access to dissolved N and P in the water column: the only way to restore the C:N:P balance and accelerate biodegradation is to add the elements that are limiting, i.e., N and P. This process, called 'biostimulation', is discussed in greater detail in Chapter 6.

The largest use of biostimulation to aid in cleaning up a major oil spill was in Prince William Sound, AK, during the 1989 *Exxon Valdez* incident (Atlas and Hazen 2011). Nutrients in various formulations were used to accelerate the biodegradation of oil that contaminated the sand and gravel beaches. Field evidence showed that biodegradation was indeed accelerated but, after a few years, both the treated and untreated sites were equivalent in terms of residual oil remaining (Pritchard et al. 1992). *It is generally true for almost all countermeasure technologies that, eventually, nature catches up with accelerated recovery and intervention does not necessarily result in greater total oil degradation, just faster rates.* However, biostimulation is not recommended for use in accelerating oil biodegradation on open water because nutrients cannot be retained effectively at high enough concentrations in the water column due to dilution. The same is true for bioaugmentation; dilution reduces the bioaugmenting culture rapidly. Use of bioremediation for oil spill cleanup likely should be based on a Net Environmental Benefit Analysis (NEBA) described in Chapter 4.

3.1.5 Salinity

Although microbial degradation of oil has been observed over the salinity range observed in the natural environment, changes in salinity may affect oil biodegradation by changing the composition of the microbial population. For example, dramatic variations in salinity may occur in estuarine environments where marine organisms mingle with freshwater species. Salinity is relevant not only to marine ecosystems but also to freshwaters, shorelines and overland spills where brine may be a co-contaminant with the oil.

Hydrocarbon biodegradation has been reported to be inhibited in non-saline environments and groundwater when salinity increases (Margesin and Schinner 2001; Minai-Tehrani et al. 2009; Ulrich et al. 2009). In contrast, most marine species have an optimum salinity range of 25 to 35and grow poorly or

[4]*Oligotrophic environments have very low concentrations of carbon and nutrients suitable for sustaining life.*
[5]*Eutrophic environments are the opposite of oligotrophic, having excess nutrients that can lead to unbalanced growth, such as algal blooms, that have subsequent deleterious effects.*

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not at all at salinity lower than 15 to 20 (Zobell 1973). In a study of hypersaline salt evaporation ponds, Ward and Brock (1978) showed that rates of hydrocarbon metabolism decreased with increasing salinity in the range of 33 to 284. Whitehouse (1984) reported an inverse relationship between salinity and solubility of PAHs. Thus, in general, a less saline environment may mean more soluble PAHs and better biodegradation rates.

Microbial mats in an Arabian Gulf environment having chronic exposure to oil spills and highly variable daily salinity and temperature fluctuations were studied by Abed et al. (2006). They reported almost complete degradation of the PAH phenanthrene and the heterocycle dibenzothiophene at a salinity of 35, and the best biodegradation of the *iso*-alkane pristane and *n*-octadecane between 35 and 80 salinity. However, inhibition of hydrocarbon biodegradation by high salinity has been reported even in environments where salt-tolerant or slightly halophilic microorganisms are dominant, such as mangroves and intertidal microbial mats. As reviewed by Martins and Peixoto (2012), Diaz et al. (2002) assessed hydrocarbon biodegradation in a mangrove microbial consortium immobilized onto polypropylene fibers. They verified that alkane biodegradation increased from <40% at 0 salinity to 65% at >40 salinity and remained there through 140 salinity, then declined back to <30% at 180 salinity, thus demonstrating an optimum salt concentration for biodegradation by these hypersaline-adapted microbes, above which biodegradation slows. In contrast, in studies with heavy crude oil-contaminated marine sediments, Minai-Tehrani et al. (2009) reported that higher salt concentrations (sediment pore water having > 10 salinity) inhibited total crude oil and PAH biodegradation. Degradation of the PAHs phenanthrene, anthracene and pyrene was greater at a salinity of 1, and fluoranthene and chrysene degradation was better at 0 salinity. *Based on the discussion above, it appears that one cannot delineate an optimum salinity range for hydrocarbon biodegradation since such activity depends on the particular adapted microbial community that is dominant in a given environment.*

3.1.6 pH

Because seawater is slightly more buffered than freshwater, the pH of seawater is generally more stable and slightly alkaline (Bossert and Bartha 1984), whereas the pH of freshwater and soil environments is more variable. Organic soils in wetlands are often acidic, as are some surface waters where the salinity is near zero, such as Muskoka in Ontario where 92% of the lakes have an alkalinity value [6] of less than 20 mg/l, which means that they are susceptible to acidification through acid deposition (Muskoka Watershed Council 2007). The pH of mineral soils is more neutral to alkaline. Most heterotrophic bacteria (those that use organic material, including hydrocarbons, for growth) favour a neutral pH (~6.5 – 7.5) (Leahy and Colwell 1990), while fungi are generally more tolerant of acidic conditions (pH <6.5). Therefore, bacteria are the dominant hydrocarbon-degrading microbes in many circumneutral environments, such as aquatic ecosystems, but fungal biodegradation may increase in importance in acid-stressed environments.

> Oil biodegradation occurs over a range of pH values, but is generally optimum at near-neutral to slightly alkaline conditions (pH 6.5-8).

3.2 Oil Impacts on Different Aquatic Environments

3.2.1 Sand and gravel shorelines

The behaviour of spilled oil on sand and gravel shoreline environments primarily depends on the properties of the shoreline, such as the porosity of the substrate, the morphology of the shoreline (slope, topographical variability, sinuosity, stability) and the energy of the waves impinging on the shoreline. Differences will also occur in the fate of oil along marine, estuarine and freshwater shorelines, largely due

[6] *Alkalinity is a measure of the capacity of an aqueous solution to neutralize acid; it is not the same as pH.*

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to differences in biological diversity and biomass, as well as dissolved oxygen and nutrient (N and P) replenishment, as discussed above.

> Interactions between oil and fine mineral particles play an important role in natural oil cleansing in marine shorelines.

This interaction of oil and fine particles reduces the adhesion of oil to intertidal shoreline substrates through the formation of oil-mineral aggregates (OMAs; Chapter 2) that are easily dispersed by tidal action and currents. More importantly, suspended OMAs (also, described as clay-oil flocculation) enhance the availability of oil for biodegradation (Figure 2.3), and thus oil biodegradation rates may be accelerated by this process (Bragg and Owens 1995; Lee et al. 1997a, b).

> Persistence of nutrients on marine beaches is dependent on tidal and wave action.

Definitive research conducted on two beaches in Scarborough, ME (Wrenn et al. 1997a, b) clarified the role of wave and tidal energetics and geomorphology on oil biodegradation. They pointed out that beaches with strong wave action will tend to more rapidly lose nutrients needed to maintain or enhance oil biodegradation and that salinity layers are created between the fresh groundwater and the tidal seawater in both high and low wave-action beaches. These tracer studies showed that the washout rate of nutrients from the oil-impacted intertidal zone is strongly affected by the wave activity of the contaminated beach. *Wave action in the upper intertidal zone causes nutrients from the surface layers of the beach to be diluted directly into the water column, resulting in their immediate loss from the bioremediation zone. Washout due to tidal activity alone, however, is relatively slow, and nutrients will likely remain in contact with oiled beach material long enough to allow effective biostimulation on low-energy beaches.* In the two beaches studied (a high and a low energy beach), the 'sandwiching phenomenon' (a layer of lower salinity water located between two higher salinity water layers) was observed, which was confirmed by Boufadel et al. (1999) in a mesocosm study. Both the mesocosm and field studies are helpful in providing guidelines for the application of bioremediation strategies, such as biostimulation based on the optimal application of nutrients on marine beaches, which are discussed later in Chapter 6.

Recommendation: The interaction between residual oil in the water with suspended particulate matter of organic origin (e.g., phytoplankton, detritus, extracellular released products) and inorganic origin (e.g., mineral fines) (OMA and clay flocs) on the environmental persistence of spilled oil (sinking, oil biodegradation rates, etc) requires further study. This is a missing point on the calculation of the oil's mass balance following a spill.

3.2.1.1 Porosity of the substrate

Higher wave exposure enhances both physical removal and weathering processes. Wave-swept rocky shores tend to recover from oil spills within a matter of months, whereas marshes may act as a petroleum sink for many years. Tidal pumping promotes oil penetration into the sediments. Carls et al. (2003) showed that tidal pumping causes dissolved hydrocarbons in beach pore waters to move laterally into streams cut through beach sediments. *On coarse-grained shorelines like cobble and sandy beaches, oil can penetrate more deeply and remain longer (when it is trapped below the limit of wave action; Figure 2.3), compared to finer grained sediments such as silt and clay*. However, oil is more easily removed from coarse-grained sediments by water flushing. Interactions of oil with tidal action, waves and shoreline substrate may also form asphalt-like oil-sediment mats that are resistant to further biological and photochemical weathering. Thus,

> The rate and depth of oil penetration depend primarily on the porosity of the substrate.

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porosity may control oxygen availability and hence control natural biodegradation rates. More detailed discussion of oil interacting with armoured beaches is presented in Chapter 2.

3.2.1.2 Shoreline morphology

> Shoreline morphology influences oil fate via effects on the nature and extent of transport of the oil both longitudinally along the shoreline and inland.

Steep shorelines (particularly if exposed to wind and wave action) will experience significant erosional processes; therefore, oil deposition would tend to develop in a narrow band within the zone of wave action where it would be subject to repeated cycles of re-suspension and deposition. Steep shorelines usually have little or no vegetative cover, thus further decreasing the likelihood that oil will remain onshore for substantial lengths of time. Shorelines with low sinuosity (i.e., running in a straight line) will tend to have sandbar formation offshore, producing the potential for oil to be deposited on the sandbars as well as along the shore itself. Shorelines with higher sinuosity (curving with embayments) will produce lower-energy conditions on lee-sides of the curvatures, where oil deposition would be more likely to occur. High topographical variability along a shoreline (e.g. rocky outcrops, stream mouths, dunes and human infrastructure, such as docks, weirs and breakfronts) will create variable exposure to wind and wave action, creating, in turn, variable oil deposition. Unstable shorelines subject to erosion will experience changes in position and composition annually, thus affecting the rate at which deposited oil is either re-suspended in the water column, pushed deeper and/or higher up the shore, or transported in bulk via major bank failures.

3.2.1.3 Wave exposure

Higher wave exposure enhances both physical removal and weathering processes. Wave-swept sand and gravel shores will often be unstable, as discussed above, resulting in a dynamic and somewhat unpredictable sequence of deposition and suspension. The combination of wind and wave exposure with low sinuosity and low stability would be the most likely to produce deposition/re-suspension cycles. The ultimate fate of spilled oil under these conditions would depend upon the type of oil and rate of weathering.

Rocky shores with high wave exposure tend to recover from oil spills within a matter of months, particularly when the shoreline has a steep slope and vegetation cover is minimal (IPIECA 1995). In the marine environment, more gradually-sloping rocky shores with heavy vegetation cover (e.g., *Fucus*, a ubiquitous brown seaweed) and tide pools may experience some stranding of oil in the upper intertidal zones. In freshwater lakes, rocky shores are often bare or limited to a lichen/moss vegetative cover; therefore, oil would be less likely to strand (depending upon the density of the lichen/moss cover).

3.2.2 Estuarine shorelines

> Estuarine shorelines are complex, dynamic and diverse; therefore, the deposition and weathering of spilled oil in these environments will be highly site-specific and difficult to predict.

Slopes are highly variable and range from gently sloping **(Figure 3. 2)** land to steep cliffs. The waterline is not constant due to tides and storm surges, creating a 'shore zone'. The variability in slope, energy climate, sediment composition and other physical characteristics (including the presence of woody debris) creates highly-variable transport and weathering conditions for spilled oil. Rates of weathering will also depend upon the nature of vegetative communities. Estuarine vegetation includes eelgrass beds, salt marshes and swamp forests. Mud flats are common. Engineered structures are common along estuarine shorelines (e.g., bulkheads and

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riprap revetments), adding to the complexity. Thus, there can be multiple stranding sites for spilled oil in estuaries.

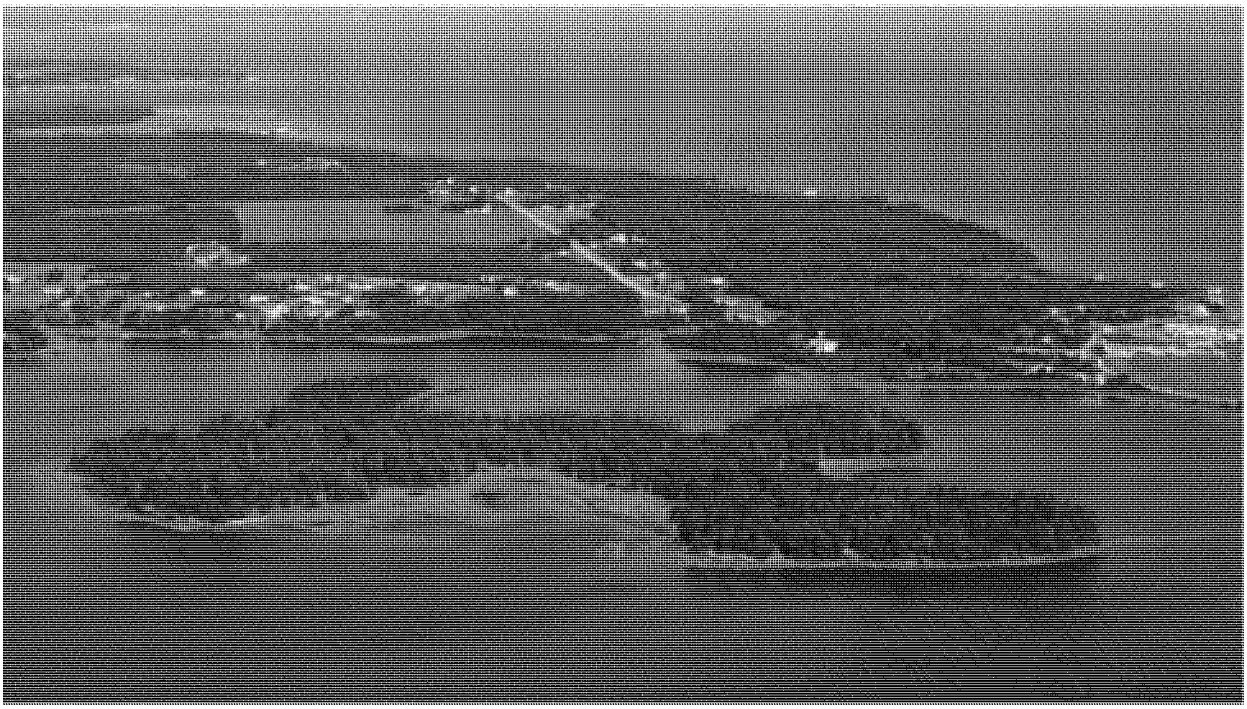

Figure 3.2 Pugwash River Estuary, Nova Scotia

3.2.3 Riverine shorelines

The behaviour of spilled oil on river shorelines will be affected by the stream order[7], seasonal hydrology, channel morphology, the tendency for hyporheic flow and the nature and extent of the riparian zone. A spill into a lower-order stream (e.g., a small headwater stream) with a steep, rocky shoreline will behave differently from a spill into a higher-order stream with a broad floodplain and lower slope. The timing of the spill relative to seasonal flow will also be an important deter minant. During the spring freshet, a spill would be rapidly transported downstream and along the upper margins of the stream channel, where it could become stranded once river flow drops. A spill that occurs during low-flow conditions (particularly when the river is fully or partially ice-covered) may tend to remain in a smaller area; however, its ultimate fate would depend on further transport and weathering processes once ice melts and flows increase. Channel morphology will affect the deposition and weathering of spilled oil via the potential for stranding on mid-channel bars or in side -channels. Higher-order streams will be more likely to have complex channel morphology. Hyporheic flow (Fig ure 2.3) along stream margins (via surface -to-near-surface groundwater connection) is common, and spilled oil could enter this flow system, then adsorb to particles along the flow -path. Hyporheic flows also occur vertically across the channel, feeding water into the hyporheic zone under the river. These downward flows m ay re -emerge immediately following an obstruction or further downstream, creating transport pathways through sediments for dissolved and particulate oil, trapping oil in sediments, and providing sources of dissolved oil to interstitial water of streambeds and bottom waters where flows re -emerge. The width and vegetative cover of the riparian zone along the river shoreline will also affect the transport and weathering of spilled oil. A spill that occurs during high -flow conditions will likely enter the ripar ian zone and could become stranded on

[7]*Order is reflection of the size (and sometimes complexity) of a river or stream network, from small (brook or stream) to large (major river)*

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vegetation or on the finer, organic substrates common to these zones. Riparian zones can contain oxbow lakes and wetlands, which could become sinks for spilled oil.

Recommendation: More research is required on the persistence of oil in riverine, lacustrine and mud flat environments. Such research would benefit risk assessments and aid response operations. Research also needs to focus on mixing and physical dispersion in high gradient, high energy rivers and hyporheic zones connected to groundwater resources.

3.2.4 Rocky cliffs and beaches

On coastlines where the land surface is at a relatively steep angle below the water table, the abrasive action of continuous marine waves may create steep cliffs, and slopes constantly being eroded. As coastal cliffs collapse, the shoreline recedes inland. The speed at which this happens depends, in particular, on the strength of the surf, the height of the cliff, the frequency of storm surges, the hardness of the bedrock and, in the Arctic, the importance of permafrost to bond shore sediments Rocky cliffs are typically not considered a high ecological risk from oil spills unless they provide habitat to wildlife. These sites are frequently left to self-clean by natural physical processes (e.g. scour by waves, etc.) as the oil is not readily absorbed into rocks.

3.2.5 Lakes

> The behaviour of spilled oil in lakes is affected by lake size, basin morphology, exposure to prevailing winds, internal wind-driven lake currents (e.g., seiches) and season of the spill.

Oil spilled into small lakes would likely rapidly spread to cover a large proportion of the lake surface. Residual oil remaining after evaporative loss in small lakes would likely be deposited along the windward shoreline, but if the lake is well protected with steep shores, some residual oil could remain in the narrow littoral zone immediately offshore. Oil spilled into larger lakes would be pushed and mixed by wind and wave action. The eventual location of residual oil after the initial evaporative loss would depend upon the nature of the windward shoreline (particularly slope and substrate size), the nature of the littoral zone immediately offshore (if there are large littoral macrophyte[8] beds, the oil could be captured within those beds), and the effect of lake currents. Seiches[9] can push residual oil up the shoreline on the windward side, where it could become stranded on vegetation or fine shoreline sediments. Oil spilled during seasonal lake turnover[10] in the spring and fall could be subject to mixing deeper into the water column where dissolved oxygen concentrations are low.

As an example of oil spilled onto a lake, on August 3, 2005, a Canadian National Railway train derailed near Wabamun Lake in central Alberta. The derailment released 712,500 L of Bunker C oil and 88,000 L of pole-treating oil from the rail cars, across the shoreline and into the adjacent Wabamun Lake. Workers removed much of the oil via vacuum trucks and by hand. Sorbent booms were ineffective in containing the Bunker C oil. Some containment booms were deployed, but insufficient numbers were available. Eventually, the size of the spill forced the engagement of a response organization with experience and equipment suitable for a larger response effort. The cleanup took years. Although the heavily-used recreational lake was re-opened a year after the spill, the Alberta government was still issuing health warnings about tar balls (Figure 2.3) and oil sheen two years later (Goodman 2009).

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[8] *Macrophytes are aquatic plants that may be emergent, submergent or floating*
[9] *Seiches are standing waves in an enclosed or partly-enclosed water body*
[10] *Turnover is a seasonal phenomenon of water movement from top to bottom according to temperature (density), more common in large, deep lakes than shallow lakes*

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To estimate the hydrocarbon -degrading capabilities of microbes naturally present in oil -contaminated Lake Wabamun sediment, laboratory tests were conducted with spilled Bunker C oil reco vered from the lake and the reference oil, Alberta Sweet Mix Blend (ASMB), in microcosms at 22 °C for four weeks or at 4 °C for eight weeks (Foght 2006; Wang et al. 2011). The results showed that Lake Wabamun sediment microbes were capable of degrading a significant proportion of the ASMB reference oil at both incubation temperatures. However, biodegradation of Bunker C was far less significant than ASMB, with smaller masses of hydrocarbons being degraded under both incubation conditions. For example, only 126 mg Total Petroleum Hydrocarbons per gram of Bunker C were degraded compared with 263 mg/g ASMB in parallel ASMB cultures (fresh sediment, 22 °C for four weeks). Under the best incubation conditions, biodegradation accounted for loss of only ~12 -13% by weight of the Bunker C oil added to the cultures. This shows that only a small proportion of LMW hydrocarbons is biodegradable with a high proportion of components (including high molecular weight components, such as asphaltenes and resins) either not readily biodegradable or not detectable by gas chromatography (Appendix A) in the spilled Bunker C oil. *This persistence is most likely due to the recalcitrance of the major constituents of Bunker C oil.*

Oil spilled into the Great Lakes would be subject to hig h-energy phenomena similar to some marine situations, but without tidal effects. Lake currents would be expected to play a major role in behaviour and persistence. An offshore spill adjacent to strong alongshore lake currents could be widely dispersed in a short period of time. *Areas that are chronically contaminated with low levels of hydrocarbons (e.g., commercial ports and recreational marinas) might be expected to harbour larger numbers of microbes adapted to oil degradation and therefore might be able to respond more quickly to an oil spill than pristine areas where biodegradation could be delayed.*

3.2.6 Wetlands

Wetlands, including marshes, swamps, fens, muskeg and mud flats, are sensitive to oil spills because of their complex and rich ecosystem diversity and their ability to collect and filter runoff from surrounding environments. Saline wetlands are common in some parts of Canada, particularly the prairies, but the behaviour of spilled oil in these systems has not yet been rigorously studied.

The physical and chemical characteristics of freshwater and saltwater wetlands that affect the dispersion and weathering of oil include small areas of shallow water, finer sediments with high organic content, high vegetation cover and high biochemical oxygen demand (leading to anaerobic conditions). Oil spilled into these systems will not be widely dispersed by wind. Rather, it will tend to be stranded on fine sediments or on vegetation. Transport out of the wetland would be via small stream discharge points.

Saline wetlands (**Figure 3.3**) are highly productive and are often ringed by broad salt flats with salt -tolerant vegetation. They are important habitats for waterfowl. Biodegradation of residual oil spilled into these systems would depend upon the presence a nd ability of salt -tolerant microbes (Section 3.1.5) to degrade the oil. In addition, if the oil becomes entrained within the anaerobic sediments commonly found within the wetland, the biodegradation rate of the hydrocarbons may be significantly reduced.

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Figure 3. 3 Aerial view of replicate plots of an experimental oil spill on a saltwater wetland in Nova Scotia, Canada, in 2001 *(joint project between DFO-Canada and U.S. EPA).*

3.2.7 Permafrost

Strategies are needed to deal with the likelihood of spills occurring in remote Arctic regions as exploration and production activities increase. The U.S. EPA (2013) published a short report on the impact of climate change on Alaskan permafrost [11]. Over the past 50 years, temperatures across Alaska have increased by an average of 1.9 °C. Winter warming was even greater, rising by an average of 3.5 °C and is projected to increase an additional 1.9 to 3.9 °C by the middle of this century. Thawing permafrost associated with climate change may cause major impacts on transportation , forests, ecosystems and the economy in the future. Over the past 50 years, thawing permafrost and increased evaporation have caused a substantial decline in the area of Alaska's closed -basin lakes (lakes without stream inputs and outputs). These surface waters and wetlands provide breeding habitat for millions of waterfowl and shorebirds that winter in lower latitudes (Karl et al. 2009). Furthermore, as temperatures rise and permafrost thaws, the softening soil can interfere with tree root systems (causing trees to sink into the ground) and infrastructure supporting oil and gas pipelines, etc. , possibly affecting pipeline integrity. *With the exception of a few recent reports (e.g., Yang et al. 2014), little is known about the impacts of oil on permafrost e cosystems and related surface meltwaters, for example, loss of permafrost slope stability due to black -body heating by solar radiation.*

<u>Recommendation</u>: Research is needed to further our understanding of the effects of spilled oil on permafrost ecosystems and how best to develop appropriate response strategies that mitigate damage without doing further harm.

The ADAPT program (Arctic Development and Adaptation to Permafrost in Transition) was designated to fund innovative Canadian research in the field of permafrost change. The ADAPT mission is to utilize this research to develop a structure of "Integrated Permafrost System Science" to understand and predict consequences of rapid changes on ecosystems and economics.

3.2.8 Ice-covered Arctic marine environments

The extent of sea ice is declining throughout the Arctic Ocean and adjacent seas . Some sea ice persists from year to year (known as perennial sea ice), often getting thicker where it forms pressure ridges (e.g., the stamukhi zones between the 10 -20 m isobaths). Over the past several decades, the perennial sea ice

[11] *Permafrost is ground that remains frozen for two or more consecutive years.*

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has declined dramatically to be replaced by seasonal (first year) ice . Ocean currents and winds have also played an important role, pushing perennial sea ice out of the Arctic basin. The average sea ice extent in September has decreased by 11.5% per decade over the past 30 years.

Oil spills in these sensitive areas as they warm through global climate change could be devastating to the changing ecosystems and exacerbate the effects of melting snow and ice.

A number of recent reviews have highlighted the results of landmark field studies conducted within the Arctic by Canadian, Norwegian and American scientists on the environmental processes that affect oil behaviour and weather ing in open water and in ice , and the effectiveness of current and emerging spill response countermeasures (Dickins 2011; NRC 2014).

In January 2012, members of the international oil and gas industry launched a major collaborative effort to enhance Arctic oil spill capabilities under the auspices of the International Association of Oil and Gas Producers (IOGP). This collaboration, called the Arctic Oil Spill Response Technology Joint Industry Programme (JIP), is aimed at expanding industry knowledge and proficiencies in Arctic oil spill response (http://www.arcticresponsetechnology.org/about-the-jip).The JIP is currently conducting a series of research projects in laboratory conditions on six key areas of research: dispersants, environmental effects, trajectory modeling, remote sensing, mechanical recovery and in situ burning.

Recommendation: In light of the Gulf of Mexico oil spill incident, a need has been identified to advance knowledge of , and ability to deal with , subsurface blowout s in ice -covered marine environments. While the National Energy Board has identified the need for additional precautionary measures in these important environments (e.g., same-season relief well capacity), further research is needed to aid in development of oil detection and response strategies.

Ecological exposure to oil from drilling causes adverse chronic effects on species composit ion and diversity. *Acute exposures from spills can be severely damaging to fisheries, marine mammals, waterfowl, corals and coastal wildlife by affecting feeding, activity, avoidance, growth and reproduction (Semanov et al. 1997). An important component of Arctic marine productivity is the under -ice community that develops in response to sunlight penetrating sea ice (Chapter 4). This community would be highly exposed to oil discharged from a subsurface blowout. Marine mammals would likewise be at high risk given that they are forced to breathe at small holes in the ice where oil would concentrate (Chapter 4).* The primary sources of oil in the Arctic marine ecosystem are from drilling activit ies and spills during transport. The risk of crude oil and/or fuel spills in the Arctic Ocean has increased with announcements in 2015 of imminent deep sea Arctic drilling in the Chukchi Sea between Alaska and Russia and the first commercial cargo shipment navigating the Canadian Northwest Passage in 2013.

Oil spilled in the marginal ice zone and polynyas [12] during high primary production periods c ould cause extensive damage, especially in shallow waters important to organisms at all trophic levels in the food chain. For example, during the summer season, runoff from the Mack enzie River forms a warm, low - salinity 'thermal bar' along the coast of Beaufort Sea. Anadromous fish species congregate in these highly productive waters to feed, grow and mature before returning to rivers in the autumn to spawn , and these regions form cr ucial summer habitat for Beaufort beluga populations; these living resources and others are depended on for food by native northerners . Onshore winds would carry surface oil into these shoreline zone s, exposing resident species to hydrocarbons at important phase s in their life cycle s. Existing response infrastructures are located at far distances, oil is less accessible when the sea is ice-covered, and conditions may be equally challenging when open water provides condit ions for large waves, storm surges and coastal inundation. Topping this off is the slow rate of biodegradation at these cold temperatures (Section 3.1.2), which could lead to persistence of the spilled hydrocarbons for

[12] *A polynya is an area of open water within the sea ice pack*

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decades. For years many scientists have agreed that the Arctic environment will require special precautions to minimize the risks of accidental oil spills (NRC 2014).

Uncertainty is high regarding the effect of oil in ice-covered environments. Oil tends to become trapped between ice floes, under ice and within brine channels, resulting in a lengthening of the contamination and exposure times (Figure 2.3). It is likely that recovery from a spill would require years for affected littoral and benthic communities to return to health.

Drilling for hydrocarbons in the Arctic demands improved methods to detect and monitor the behaviour of spills on, in and under ice and within the water column in the event of a subsurface blowout. In addition, oil interactions with permafrost and spring melt are important unknowns in terms of access to sites for spill response, rapid spreading of oil during freshet, slow rates of weathering and how hydrocarbons interact with ice and suspended particulate matter.

In pack ice, oil accumulates at the water surface, flowing around any ice present. Oil can also collect under the ice, which usually has rough topography associated with pressure ridges. At sea-ice concentrations less than 30%, oil behaves as in open water (Venkatesh et al. 1990), but at higher ice concentrations, the oil drifts with the ice. The equilibrium oil thickness in slush or brash ice [13] is nearly four-fold higher than on cold water. As a result, the oil-contaminated area at higher ice concentrations is several orders of magnitude smaller, but the oil layer is thicker. Thus, the presence of pack ice significantly slows oil spreading. *This means that a smaller area is affected, but the impact on that area is greater and may overwhelm the microbial community. Recent research (Greer et al. 2015) has discovered the presence of oil-degrading microbial biofilms at the bottom of sea ice, but the prevalence and persistence of such structures is unknown.*

As it weathers, the fate of oil in ice is affected by density, viscosity, surface and interfacial tensions and water content. As described in Chapter 2, typical weathering processes include evaporation, dissolution, emulsification with water to form water-in-oil gels, photodegradation and dispersion. Singsaas et al. (1994) found that the rates of evaporation, dispersion and emulsification were retarded in pack ice. They reported that the primary factors that reduced the weathering rate when compared to open water and temperate conditions were wave damping, limited spreading of oil bounded by sea ice and low temperature.

Immediate surface evaporation results in losses of small alkanes (C_5-C_{10}) and monoaromatic compounds, such as BTEX (benzenes, toluenes, ethylbenzene and xylenes). Several experiments with refined gasoline and diesel (Serova 1992; Ivanov et al. 2005) have found nearly complete evaporation on the surface of ice during the summer season. The highest evaporative loss occurs in the presence of uncompacted snow, while increasing snow density and thickness reduces evaporation (Belore and Buist 1988).

To determine the importance of dissolution, Faksness and Brandvik (2008a, b) subjected six oils (five crude oils and a heavy fuel oil) to freezing in sea ice during the winter and quantified the migration of water-soluble components from the oil into the ice. They found that the content of water-soluble compounds in the ice decreased with ice depth, and the concentrations changed over time, confirming that the water-soluble components had been transported from the overlying oil through the brine channels in the ice to the underlying water. The field experiments showed that both ice thickness and air temperature prior to an oil spill are important for the distribution of water-soluble components in ice.

Moussification, or the formation of water-in-oil emulsions, forms by water-drop entrainment during mixing (Figure 2.5). This is an important weathering process both in cold and warm water. Emulsification increases the volume of the slick and reduces the biodegradation rate. *The formation of stable water-in-oil*

[13] *Brash ice is an accumulation of the wreckage of other forms of ice made up of fragments not > 2 m across.*

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emulsions hampers various oil spill cleanup efforts by increasing volume for containment, lowering combustibility for in situ burning, and increasing viscosity so that chemical dispersion is more difficult.

Photooxidation (Chapter 2) and biodegradation are important transformation processes that occur with petroleum products released into the marine environment (Garrett et al. 1998; Dutta and Hara yama 2000; Prince et al. 2003). Whereas photooxidation is highly seasonal in the Arctic, biodegradation rates should be relatively constant in Arctic sea water. Garret et al. (1998) demonstrated that *photooxidation and biodegradation operate on different aromatic hydrocarbon components in crude oils*. Whereas biodegradation and evaporation/dissolution result in the depletion of unsubstituted aromatic compounds relative to their alkylated homologues, photooxidation selectively attacks the alkylated aromatic compounds (Prince 1993; Prince and Clark 2004).

Finally, with respect to sedimentation (Figure 2.3), most oils do not sink as their specific gravity is less than that of seawater. However, when suspended inorganic particulate matter interacts with oil, the conglomerate formed may be negatively buoyant. This natural process may occur in near-shore environments with a high suspended particle load such as that associated with glacial tills in the Arctic. Oil sedimentation is usually considered to be disadvantageous for the overall removal of oil from the environment because oil in the sediment phase tends to be limited by oxygen supply, and anaerobic biodegradation of hydrocarbons is a slow, poorly understood process (Section 3.1.3), particularly when combined with low temperatures.

3.2.9 Offshore Surface Spills at Sea and in Large Freshwater Lakes (e.g., Great Lakes).

3.2.9.1 Spreading

The spreading of oil on water is one of the most important processes during the first hours of a spill, provided that the oil pour point (the temperature at which it becomes semi-solid and loses its flow characteristics) is lower than the ambient temperature. The principal forces influencing the spreading of oil (Figure 2.3) include gravity, inertia, friction, viscosity and surface tension. These processes increase the overall surface area of the spill, thus enhancing mass transfer via evaporation, dissolution and later biodegradation. Oil accumulated at the water surface can be pushed by the wind, generally slightly to right of the wind vector in the northern hemisphere. Wind, currents and tides, the three main contributors to transport of oil following an oil spill, may be used to project where the oil might go and what it might encounter in its path.

> At Arctic temperatures, the presence of snow and ice can substantially affect the spreading of oil.

3.2.9.2 Evaporation

Evaporation (Figure 2.3) is responsible for the removal of a large fraction of the oil including the more acutely toxic, lower molecular weight components. For oil on water, evaporation removes virtually all the normal alkanes smaller than C_{15} within one to ten days. Gros et al. (2015) reported > 50% of $\leq C_{17}$ hydrocarbons disappearing within 25 hours from an experimental 4,300 L North Sea oil slick of < 10 km^2 area and < 10 µm thickness. For oil sheen, > 50% losses of $\leq C_{16}$ hydrocarbons were observed after 1 hour. Volatile aromatic compounds, such as the monoaromatics benzene and toluene, can also be rapidly removed from an oil slick through evaporation. However, these volatile oil components may be more persistent when oil is stranded in sediments. The volatile components make up 20-50% of most crude oils, about 75% of No. 2 fuel oil, and about 100% of gasoline and kerosene. As a result, the physical properties of the remaining slick change significantly (e.g., increased density and viscosity). Major factors

> In terms of environmental impacts, evaporation is the most important weathering process during the early stages of an oil spill.

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influencing the rate of evaporation include composition and physical properties of the oil, wave action, wind velocity and water temperature (Jordan and Payne 1980). Evaporation leaves behind the heavier components of the oil, which may undergo further weathering or may sink to the ocean floor. For example, spills of lighter refined petroleum-based products, such as kerosene and gasoline, contain a high proportion of flammable light ends, which may evaporate completely within a few hours, thereby reducing the toxic effects to the aqueous environment. Heavier oils leave a thicker, more viscous residue, which may have serious physical and chemical impacts on the environment.

3.2.9.3 Dissolution

Although dissolution (Figure 2.3) is less important than evaporation from the viewpoint of mass loss during an oil spill, dissolved hydrocarbon concentrations in water are particularly important due to their potential influence on the success of bioremediation and the effect of toxicity on biological systems. Light aromatic compounds, such as benzene and toluene, are the most soluble in seawater, and the most acutely toxic. However, these compounds are also first to be lost through evaporation, a process which is 10-100 times faster than dissolution, and oil contains only small amounts of them which tends to mitigate dissolution. Even so, the impact of LMW aromatics on the environment is much greater than simple mass balance considerations would imply (NRC 1985). Dissolution rates are also influenced by photochemical and biological processes. The larger components of oil (4-6 ring PAHs, alkanes >C_{20}, etc.) are quite insoluble in aqueous solution, so they tend to persist longer in the environment.

> The extent of dissolution depends on the composition and state of the oil and occurs most quickly when the oil is finely dispersed in the water column.

3.2.9.4 Emulsification

Emulsification (Figures 2.3 and 2.5), or formation of oil-in-water emulsions, involves the incorporation of small droplets of oil into the water column, resulting in an increase in surface area of the oil. Emulsification occurs by physical mixing promoted by turbulence at the sea surface. The emulsion thus formed is usually very viscous and more persistent than the original oil and is often referred to as mousse because of its appearance. The formation of these emulsions causes the volume of oil to increase three- to four-fold, which slows and delays other processes that would allow the oil to dissipate. Conventional oils with an asphaltene content greater than 0.5% tend to form stable emulsions that may persist for many months after the initial spill has occurred (WSP Canada, Inc. 2014). *However, as reported in Chapter 2, dilbits do not emulsify as easily as conventional oils. Dilbits, if spilled into turbulent water, can form entrained or unstable emulsions, which break down into water and oil within minutes or a few hours at most, once the sea energy dissipates (Fingas 2015).* Oils containing a lower percentage of asphaltenes are less likely to form emulsions and are more likely to disperse.

> An emulsion is formed when two liquids combine, with one becoming suspended in the other.

<u>Recommendation</u>: Research is needed to further our understanding of the properties of dilbit that preclude it from being less subject to emulsification than other crude oils.

In general, oil-in-water emulsions are not stable, but water-in-oil emulsions are. Water-in-oil emulsions containing 50 to 80% water are most common and have a reddish-brown colour and grease-like consistency. Such emulsions are highly resistant to biodegradation. However, oil-in-water emulsions can be maintained by continuous agitation, interaction with suspended particulates, and the addition of chemical dispersants, discussed further in Chapter 6. Chemical dispersion may influence oil biodegradation rates by increasing the contact area between oil and microorganisms or by increasing the

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dissolution rates of the more soluble oil components. Emulsification is less likely to occur in fresh water, even for spills of heavier oils, because significant mixing energy often does not persist long enough to generate the stable emulsions seen at sea. Moreover, brine composition (alkalinity in particular because of a buffering effect) is int imately tied to the pH in determining the stabilizing properties of the interfacial films (Strasser 1968). Brines with high $[Ca^{+2}]$ and a high Ca^{+2}/Mg^{+2} ratio form nonrelaxing, rigid films around the water droplets, resulting in stable emulsions (Jones et a l. 1978). Higher concentration of divalent ions and high pH result in reduced emulsion stability. Where emulsions do occur, they may separate into oil and water again if heated by solar radiation under calm conditions or when stranded on shorelines.

3.2.9.5 Photooxidation

Oxidation is promoted by sunlight (Figure. 2.3) and the extent to which oxidation occurs depends on the type of oil and the form in which it is exposed to sunlight. However, this process is very slow and , even in strong sunlight, thin films of oil break down at no more than 0.1% per day. The formation of tar balls is caused by the oxidation of thick layers of high viscosity oils or emulsions. This process, also called aggregation, forms an outer protective coating containing heavy compounds, result ing in an increased persistence of the oil in the environment. Tar balls, which are often found on shorelines and have a solid outer crust surrounding a softer, less weathered interior, are a typical product of this process. Oil aggregates may exist from a month to a year in enclosed seas and up to several years in the open ocean. They complete their cycle by slowly degrading in the water column, on the shore if washed the re by currents or (if they lose buoyancy) on the sea bottom.

> In sunlight, oils react chemically with oxygen either breaking down into soluble products or forming persistent compounds called tars.

3.2.9.6 Sedimentation/sinking

Since seawater has a specific gravity of approximately 1.03, only heavy oils are dense enough, or weather sufficiently, to result in sinking in the marine environment (Figure 2.3). Sinking usually occurs due to the adhesion of sediment particles and/or dense organic matter to the oil. Shallow waters are often laden with suspended solids (such as planktonic organisms or clay/silt particles) providing favourable conditions for sedimentation. This is especially evident in estuaries, which are c haracterized by having relatively high concentrations of suspended particulate matter that aid in the sedimentation of oil droplets (NRC 2003). Valentine et al. (2014) reported a 3,200 km^2 region in the vicinity of the Macondo well contaminated with excess hopane attributed to deposition from the Deepwater Horizon (DWH) blowout. The pattern of deposition was consistent with both a 'bathtub ring' of oil impinging on the continental slope and a fallout plume of dirty marine snow where suspended oil particles sank to the sediment. Another report from the DWH spill by Gutierrez et al. (2013) implicated the role of exopolysaccharides from enriched strains of hydrocarbon degraders in contributing to the formation of oil aggregates during the spill (Figure 2.3). Oi l stranded on sandy shorelines often becomes mixed with sand and other sediments. If this mixture is subsequently washed off the beach back into the sea it may then sink. In addition, if the oil catches fire after it has been spilled, the residues that sometimes form may be sufficiently dense to sink.

> Some heavy refined products have a specific gravity ≥ 1.0 and so will sink in fresh or brackish water.

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3.3 Importance of Oil Type in Relation to the Environment

3.3.1 Light oils (diesel, heating oil, light crudes)

These types of oil (Section 2.3.2) are characterized as being moderately volatile and leave a residue (up to one-third of spill amount) after a few days. They contain moderate concentrations of highly toxic (soluble) compounds, especially distilled products. They may result in long -term contamination of intertidal resources and have the potential for acute subti dal impacts (dissolution, mixing and sorption onto suspended sediments). Cleanup can be very effective.

3.3.2 Medium crude oils (most crude oils)

These oils (Section 2.3.3) will evaporate moderately within 24 hours, have a maximum water soluble fraction of 10-100 ppm, can cause severe and long -lasting contamination of intertidal areas, and can be highly toxic to waterfowl and fur-bearing animals. They are chemically dispersible and can be cleaned up effectively if response is fast.

3.3.3 Heavy oils

Heavy oils (Sec tions 2.3.7) include bitumen and dilbit and refined fuel oils such as No. 6 fuel oil and Bunker C. They do not readily evaporate or dissolve in aqueous solution (water soluble fraction is <10 ppm), can contaminate intertidal areas heavily with long -lasting effects, cause severe impacts to waterfowl and fur-bearing mammals through coating and ingestion, and can sink readily into sediments, exerting adverse effects on benthic communities. *They weather slowly and are difficult to chemically disperse. Shoreline cleanup is difficult under all conditions.*

A major report was published in 2013 by the Canadian government that provided a comprehensive, in -depth review of the properties, composition and marine spill behaviour, fate and transport of two diluted bitumen products from the Canadian oil sands, namely, Access Western Blend (AWB) and Cold Lake Blend (CLB). These two products were chosen to represent all dilbits in use in Canada because they were the highest volume products transported by pipeline in Canada in 2012 and 2013 (Government of Canada 2013). The report documented results from laboratory and wave tank studies of the properties and composition of fresh and weathered AWB and CLB. Included were measurements of the potential of the two products to evaporat e, photo-oxidize, and sink in saltwater, as well as how well the dilbits behaved when treated with dispersants. The major results of the in-depth investigation are summarized below:

- Both dilbit products floated on seawater even after exposure to visible li ght and mixing with water;
- The dilbits sank when they came in contact with fine particulate matter in seawater and mixed by high energy wave action;
- When Corexit 9500 was used in an attempt to disperse the dilbit in seawater mixed by high energy breaking waves, little to no dispersion occurred even when fine particulates were added to aid the dispersion; and
- The two dilbit products did not evaporate as well as conventional crudes and fuel oils, but their behaviours were similar in other ways, such as emulsi on formation and how they in teracted with sediments.

It should be noted that dispersant testing in the se wave tank studies w as conducted at only one temperature, 8.3 °C. At that temperature, dispersants are less effective against heavy fuel oils than at higher temperatures (Srinivasan et al. 2007). Government of Canada (2013) did recognize this restriction in its conclusions: *"The physical properties (e.g., density, viscosity and adhesiveness) of these products limit the effectiveness of currently -available spill treating agents, thus restricting remediation and*

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potentially contributing to the persistence of the products in marine environments where seawater temperature is < 8 °C."

<u>Recommendation:</u> More research is urgently needed to determine the highest-risk combination of environmental characteristics when dealing with dilbit. Based on current knowledge and logic, for example, under what circumstances is it most likely that a dilbit product would sink? Under what conditions would weathering of dilbit be the slowest?

More research is needed to further our understanding of the behaviour of dilbit and other heavy fuel oils in a range of environmental settings, including near-shore marine, offshore marine, estuarine, freshwater lakes, rivers and wetlands, and environmental conditions, including combinations of climate, water chemistry and biological communities. This new information would help to optimize the selection of current spill response strategies and to support the development and validation of emerging technologies, including biodegradation of mousse, water-in-oil and oil-in-water emulsions. Details on the factors that influence moussification are needed for a better understanding of its properties and its treatment. The same needs pertain to tar balls.

Research is needed to further our understanding on why dilbit is less subject to emulsification than other crude oils. Knowledge gained might help in developing response strategies for all heavy oils.

3.4 Summary of Research Needs and Recommendations

3.4.1 High Priority Research Needs

3.4.1.1 Arctic Ecosystems

Research is needed to further our understanding of the effects of spilled oil on permafrost ecosystems, where challenges include limited site access, rapid spreading of oil during freshet, slow rates of weathering, and limited knowledge on the interaction of oil with ice and suspended particulate matter. New research would show how best to develop appropriate response strategies that mitigate the damage without causing further harm. In addition, little has been done to advance our knowledge of how to deal with subsurface blowouts in ice-covered marine environments. Similarly, with the recent re-invigoration of drilling for oil over the Arctic Ocean shelves and slopes, improved methods will be needed to detect and monitor the behaviour of spills on, in and under ice and within the water column in the event of such a subsurface deep sea blowout. While the National Energy Board has identified the need for additional precautionary measures (e.g., same-season relief well capacity), a number of laboratory studies have been initiated to further research, but field trials are needed to aid in development of oil detection and response strategies in these important environments.

3.4.2 Medium Priority Research Needs

3.4.2.1 Persistence of Oil in Non-Marine Environments

Most literature on oil persistence concerns marine environments. More information on oil persistence in riverine, lacustrine and mud flat environments, as well as sediments, would be beneficial for risk assessments and response operations. Although much progress has been made over the last decade in our knowledge of oil biodegradation, anaerobic biodegradation of hydrocarbons is less understood. This is important to oil that has sunk into the sediment, whether it be riverine, lacustrine, estuarine or marine, where it could persist for decades.

3.4.2.2 *High Risk Characteristics and Treatability of Dilbit*

Although the Canadian government has already begun work on the treatability of dilbit, more research is needed to rapidly determine the highest-risk combination of environmental characteristics when dealing with dilbit. Based on current knowledge and logic, for example, under what circumstances is it most likely that a dilbit product would sink? Under what conditions would weathering of dilbit be the slowest? Related to that, the need exists to further our understanding of the behaviour of dilbit and other heavy fuel oils in a range of environments, including near-shore marine, offshore marine, estuarine, freshwater lakes, rivers and wetlands under various combinations of climate, water chemistry and biological community conditions. New information is required for contingency plans to optimize the selection of current spill response strategies and to support the development and validation of emerging technologies, including biodegradation of mousse. Details on the factors that influence moussification are needed for a better understanding of its properties and its treatment. The same needs pertain to tarballs.

3.4.3 *Long Term Research Needs*

3.4.3.1 *Oil-Suspended Particulate Matter Aggregates and Dilbit Emulsification*

The interaction between residual oil in the water with suspended particulate matter of organic origin (e.g., phytoplankton, detritus, extracellular released products) and inorganic origin (e.g., mineral fines) on the environmental persistence of spilled oil (sinking, oil biodegradation rates, etc.) requires further study. Formation of aggregates is presently neglected in the calculation of the oil mass balance following a spill. Research is needed to further our understanding of the properties of dilbit that preclude it from being less subject to emulsification than other crude oils. Knowledge gained might help in developing response strategies for all heavy oils.

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CHAPTER 4: OIL TOXICITY AND ECOLOGICAL EFFECTS

Abstract

Chapter 4 reviews the literature on the toxicity of oil to marine and freshwater species to identify the primary toxic effects of oil, the mechanisms of toxicity, factors that affect measured or predicted toxicity, the ecological impacts of oil spills, and gaps in knowledge that obstruct the understanding of oil spill impacts and how to manage them.

Spilled oil can be rapidly lethal to fish, birds, mammals and shoreline organisms due to the readily dissolved components of oil and the physical effects of smothering and destruction of the thermal insulation and buoyancy provided by fur and feathers. Chronic and sublethal effects are associated with the less soluble components of oil such as the polycyclic aromatic hydrocarbons (PAHs), and some effects may be expressed long after brief exposures. Biomagnification of hydrocarbons in food webs is not an issue because fish, birds and mammals can quickly metabolize and excrete petroleum hydrocarbons, although metabolism of PAHs often creates metabolites more toxic than the parent compound.

Data on the toxicity of oil to aquatic species underpin ecological risk assessments (ERAs) of potential oil spills and environmental impact assessments (EIAs) of actual spills. The literature on oil toxicity is rich, but uneven. The most commonly reported data are from laboratory tests of oil toxicity to selected aquatic species, particularly embryonic life stages that are most sensitive to oil. However, not all data are useful. Many studies, particularly those with algae and plants, were designed to answer highly-focused questions about specific oil spills, and it is often difficult to draw general conclusions that can be applied to new circumstances.

ERAs and EIAs also rely heavily on modeling to predict the acute and chronic toxicity of hydrocarbon mixtures from chemical properties such as lipid solubility (e.g., the Target Lipid Model or TLM). The advantages of modeling are very compelling, considering the cost and time needed to run toxicity tests for every type of oil that might possibly be spilled, or every water sample collected during an oil spill. However, statistical error limits about model predictions are rarely quantified or reported in ERAs and EIAs, despite uncertainty due to untested model assumptions about the stability of oil concentrations in water, the transformations of hydrocarbons by photooxidation or metabolic oxygenation, single or multiple mechanisms of hydrocarbon toxicity, the presence of oil in different phases, additive interactions and the relationship between acute and chronic toxicity.

The toxicity of oil to aquatic species depends on the extent of exposure to the toxic components of oil. In general, light oils rich in low molecular weight narcotic compounds, such as benzene, toluene, ethyl benzene and xylene are more acutely lethal than medium and heavy oils. Heavy fuel oils with high proportions of 3- to 5-ringed alkyl PAHs are more chronically toxic than light oils due to the disruption of embryonic development by alkyl PAHs. The concentrations of polar and heterocyclic compounds in toxicity test solutions are rarely measured, so that potential effects unique to these compounds (e.g., endocrine disruption) are not routinely assessed. Recent spills of diluted bitumen (dilbit) have raised concerns about toxicity because it is often described as a 'dirty oil'. Nevertheless, only minor fish kills have been caused by diluted bitumen (dilbit) spills, and a single report of dilbit toxicity demonstrated that it causes the typical signs of chronic embryo toxicity at concentrations consistent with its alkyl PAH content. However, there are too few data on toxicity to state with certainty that dilbit is more or less toxic than conventional oils. Dilbit does differ from other oils in its behaviour when spilled, with a more rapid loss of volatiles during weathering and a greater propensity to sink in fresh water. Thus, impacts of spills may represent the unique interactions between the environmental behaviour of dilbit and the exposures of species that select sediment habitats where dilbit accumulates.

ERAs and EIAs may underestimate the potential impacts of oil spills because the relationships between oil exposure and toxicity are not always clear. Analyses of oil solutions in lab experiments and field surveys often combine dissolved oil and particulate oil in one measurement. However, the role of oil droplets as an exposure pathway is ill-defined, and there are significant analytical challenges in discriminating droplet and dissolved oil fractions. Oil concentrations in water change rapidly, and the spatial and temporal characterization of hydrocarbon concentrations in toxicity tests and during oil spills is often inadequate. In coastal and freshwater ecosystems, little is known about the interactive toxicity of oil with chemicals from municipal, industrial and agricultural effluents. Likewise, the interactions between oil toxicity and susceptibility to infection by pathogens are not well understood. As a consequence the impacts associated with an oil spill may be greater than expected.

Although, oil spills on land and to fresh waters are frequent in Canada, too little is known about the fate and effects of oil spilled to freshwater ecosystems. Most information about impacts is derived from marine spills assuming that oil fate, behaviour and effects are similar in both systems. Although the scale of oil spills from trains and pipelines is much smaller than spills from supertankers, the relative impacts may be as great or greater. There is less scope for dilution and degradation of oil in freshwater ecosystems, so that higher concentrations of oil in water may be maintained for longer periods. Shorelines and sediments of freshwater systems are always in close proximity to floating oil, which may accumulate in thick layers, and there is little time to react before spilled oil contaminates sensitive shorelines or aquatic species. Freshwater species are less able to avoid spill sites, and migrating or spawning species may congregate at high densities where oil accumulates. The fate of much of the oil spilled to freshwater ecosystems is often unknown, which raises questions about potential interactions of surface oil with sediments and groundwater flows out of and into streams and lakes (hyporheic flows). Contamination of sediments contributes to the exposure and effects on aquatic species that typically inhabit sediments, particularly the early life stages of fish that spawn in sediments.

In both marine and freshwater systems, oil spill cleanup can be as damaging as the spilled oil. Habitat damage is caused by the removal of oiled vegetation from the water or from riparian lands, clearing of vegetation to permit access to oiled sites, soil compaction by heavy machinery, erosion of shorelines and siltation and de-stabilization of streams by removal of oiled woody debris. Habitat damage can reduce the abundance and productivity of species using the habitat and allow the invasion of species tolerant of the new conditions. Shoreline erosion, once started, may propagate with high water levels or storm action, so that damage is ongoing and increasing in severity.

Chemical dispersion of marine oil slicks reduces the amount of oil contacting shorelines and surface species such as aquatic birds. Dispersion increases the rate at which oil is removed from the surface, is diluted and is available for microbial degradation. However, the dispersants may increase the impact of oil spills by increasing the exposure of subsurface species. In the top 10 m of water, concentrations of dispersed oil may be increased to levels toxic to fish and invertebrate embryos. While the intent of oil dispersion is to dilute the oil to concentrations below toxicity threshold limits, even brief exposures can cause delayed effects that are evident in the weeks, months and years following a spill. The net environmental benefits of dispersion represent the trade-offs between protecting easily-identified surface resources and much-less-obvious sub-surface resources and coastal/near-shore environments. Reports that dispersants and oil cause synergistic toxicity are incorrect. Dispersion simply increases the concentration of oil to which organisms are exposed. Dispersants themselves are moderately toxic to aquatic species when free in solution, but appear to be unavailable and non-toxic when mixed with oil. Thus, dispersant toxicity will depend on how accurately it is applied to oil, whether it is completely mixed with oil, and the concentration of 'free' dispersant in surface water if applications by spraying from aircraft miss the target. The environmental impacts of subsurface dispersant applications at the 2010 Gulf of Mexico DWH blowout are still being evaluated in ongoing studies. In Canada, there are concerns about the potential impacts of chemically-dispersed oil on east coast fisheries. Evidence–based net environmental benefit

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analysis is essential to support decisions about appropriate oil spill response options, including dispersants.

Although it is clear from laboratory studies that oil is highly toxic to aquatic species, oil spills are often followed by years of debate on actual impacts. Cause-effect relationships are often obscured by a lack of baseline data on affected species and by a high variability in population characteristics and productivity. Variations in oil exposure and effects on a given population are associated with movements into and out of affected areas, avoidance of oil exposure, fisheries closures, delayed effects of toxicity and density-dependent changes in reproduction and survival. Cause-effect is also obscured when chemical, biochemical or molecular indicators of oil exposure and effects in live or dead specimens are not included in monitoring programs. Without these indicators, it is difficult to eliminate alternative causes of disease and mortality. Post-spill monitoring is often perfunctory and ecosystem recovery is defined by how quickly organisms re-colonize spill sites and not by how quickly ecosystem function and productivity are restored. Overall, there is a lack of solid epidemiological sampling designs and a potential bias towards 'false negatives', i.e., concluding that oil spills caused no or mild effects on aquatic species, when in fact important effects occurred but were not detected. Impact on human use of natural resources are also not considered in a systematic way, and there is growing interest in assessing oil spill impacts on 'ecosystem services', i.e., *"the benefits provided by ecosystems to humans that contribute to making human life both possible and worth living"*.

Oil exploration and development in northern and Arctic regions has raised concerns about the potential for unique sensitivities of Arctic species to oil spills. Based on a limited array of toxicity comparisons, Arctic fish and invertebrate species do not appear any more or less sensitive than more temperate species when tested at equivalent temperatures. Thus, data from temperate species could be used for a first approximation of risks of oil spills to Arctic ecosystems. However, the behaviour and fate of oil in Arctic ecosystems can be strongly affected by low temperatures, ice cover and the interactions of Arctic species with ice. For example, marine mammals are forced to breathe at small gaps in ice sheets, where floating oil may accumulate. Differences in oil spill impacts between Arctic and temperate regions may be driven more by factors that affect the exposure of organisms to oil than by differences in sensitivity to oil

The capacity to identify, measure and manage a wide array of potential effects of oil spill is often limited by a paucity of appropriate tools and data. This chapter provides a series of specific research needs throughout the text, and these needs are summarized as follows.

Research is needed:

- At 'spills of opportunity' and experimental oil spills to identify effects on aquatic species at the level of populations, communities and ecosystems due to the acute and long-term toxicity of spilled oil;
- On how the behaviour and fate of fresh and weathered oil in different ecosystems interacts with the habitat selection and unique life history traits of aquatic species to control species-specific oil exposure and toxicity. These interactions are particularly important for diluted bitumen (dilbit) relative to conventional oils, and northern and Arctic ecosystems relative to temperate ecosystems;
- To understand how different oil types (e.g., dilbit) and spill-control agents (e.g., chemical dispersants) affect aquatic species under different environmental conditions, including interactions with contaminants from municipal, industrial and agricultural effluents;
- On mechanically- and chemically-dispersed oil to determine how oil droplets affect the exposure and toxicity of oil to different species and life stages of aquatic organisms. These studies include a need to develop analytical methods to reliably discriminate droplet oil from dissolved oil. Studies on the distribution and effects of dispersed oil under exposure conditions similar to those that may be encountered during response operations (including subsurface injection of dispersants) are of particular importance;

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- On strategies and protocols to recover spilled oil that avoid or minimize habitat damage, methods to restore damaged aquatic and riparian habitats and models to define the balance between the environmental costs of natural remediation and oil spill cleanup;
- On methods for finding and measuring spilled oil in different compartments of freshwater and marine ecosystems, for measuring exposure and toxicity of oil to aquatic species *in situ*, and for assessing the structure and function of ecosystems that reflect the extent and time to recovery following oil spills;
- On the contamination of ground water due to soil contamination and the entrainment of oil droplets into bed sediments of rivers by surface water-ground water exchanges;
- To develop standardized oil toxicity test methods for all classes of aquatic organisms to enable reliable comparisons of toxicity among oils, species and different environmental conditions (e.g., salinity, temperature), including rapid changes in oil concentrations in water; and
- To refine models that predict acute and chronic toxicity to aquatic species, particularly to assess the uncertainty caused by violating the assumptions underlying these models and to reduce the variance of model predictions.

Introduction

A primary concern of oil spill responders is to understand and limit the effects of spilled oil on ecosystems, natural resources and the economies and communities that depend on them. These concerns are the main drivers of prospective or retrospective ecological risk assessments (ERAs) or environmental impact assessments (EIAs) which are conducted to understand and mitigate the impacts of future developments (e.g., oil pipelines) and to investigate the effect of oil spills or chronic oiling from urban or industrial sources. Knowledge of the ecological impacts of spilled oil is derived from observational studies following actual spills and experimental laboratory studies, primarily toxicity testing with individual species. While laboratory studies of oil toxicity to species of invertebrates, fish and birds are abundant and detailed, research on oil spill effects on marine mammals, populations of organisms and ecosystem structure and function is often sparse or non-existent. The unevenness of understanding is due to differences in the questions being asked and the logistical difficulties and costs of conducting larger-scale research, including outdoor mesocosm experiments and full-scale field experiments.

The nature of concerns regarding oil spills is also changing. Past perceptions of oil spills were based on high-profile shipping disasters, where large volumes of oil were spilled with widespread acute effects on wildlife and shoreline oiling. Perspectives have changed due to the extensive research following the 1989 *Exxon Valdez* oil spill (EVOS) in Prince William Sound, AK. There is now a greater awareness of the potential persistence of oil, the ongoing exposure of aquatic species due to their behaviour and life habits, and the delayed and cumulative effects of exposure to oil and to other stressors. Concerns are growing for the potential effects of spills in extreme environments as exploration and development expand into Arctic ecosystems and into very deep offshore waters. Similar concerns relate to potential impacts during extreme seasonal conditions (winter, flooding, droughts, etc.) and impacts in remote wilderness areas that are not readily accessible for spill response. The expansion of mid-continent oil production in North America has highlighted the increasing frequency of freshwater oil spills and the question of whether knowledge of marine spills can be applied directly to managing spills to fresh waters. Recent spills of ultra-light oil, heavy fuel oils (HFOs) and diluted bitumen raise questions about the fate, behaviour and impacts of oil with extreme properties and whether the understanding of conventional oils can be applied successfully to predicting impacts of these oils when spilled.

This chapter provides an overview of what is known about oil toxicity and ecological impacts of oil spills to aquatic environments. The discussion of terrestrial effects is limited to the water-land interface, i.e., to susceptible shorelines, including salt marshes or wetlands where spilled oil is blown ashore, and riparian lands of rivers and lakes where spilled oil may be deposited by flooding. For shorelines and riparian lands, there are also concerns about potential habitat destruction by the oil spill response itself. This

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chapter is based on several comprehensive reviews (Engelhardt 1983; Leighton 1993; NRC 2005; Hodson et al. 2011; Lewis and Pryor 2013; Beyer et al. 2015; Logan et al. 2015) to avoid repeating summaries that have already been published. Information from these reviews is presented in a synthesized form but, where appropriate, some original papers are explained in detail to illustrate a point. Where still valid, research recommendations from these reports are repeated. Data from primary publications are summarized in **Appendix D, Table D.1**, which was first assembled by Hodson et al. (2011), enlarged by Logan et al. (2015), and enlarged and re-formatted for this review. The numbers of citations to original papers reflects the upsurge of interest following the 2010 Deepwater Horizon (DWH) oil spill, and this report is heavily weighted towards recent studies with fish, the most numerous in the literature.

4.1 Effects of Oil on Aquatic Organisms

The effects of oil on aquatic species start with exposure to oil or its components. Exposure may be derived from:

> **Biomagnification** – a food chain or food web phenomenon whereby a substance or element increases in concentration at successive trophic levels. Biomagnification occurs when a substance is persistent and is accumulated from the diet faster than it is lost due to excretion or metabolism.

- External coatings of oil, particularly in air-breathing species that must surface frequently and cannot avoid contact with surface slicks of oil;
- Inhaled aerosols of particulate oil and volatile hydrocarbons by air-breathing aquatic turtles, birds and mammals that contact surface oil (Engelhardt 1983);
- Oil ingested by aquatic birds and mammals that groom after being contaminated, that forage on beaches or intertidal zones contaminated by oil, or that ingest oil associated with algae or with organic or inorganic particulates (Engelhardt 1983; NRC 2005). Although invertebrates do not metabolize or excrete petroleum hydrocarbons quickly, and can contribute to the dietary exposure of predators, petroleum hydrocarbons do not typically biomagnify in food webs. *Unlike persistent chlorinated compounds, most hydrocarbons can be readily metabolized and excreted by fish, bird and, mammals. However, rates of uptake, metabolism and excretion will vary among species and with environmental factors, such as temperature.*
- The dissolved components of oil that partition from water to lipids across respiratory membranes of aquatic species that breathe via gills (invertebrates, fish), and across cell membranes of microbes and algae.

The toxicity of oil to each species varies with exposure, and exposure is a function of oil type, environmental conditions and the life history and physiology of each species. Details about exposure and factors controlling the rates of exposure for different species will be discussed in the reviews of oil toxicity that follow.

4.1.1 Immediate effects of an oil spill

The first and most obvious effect of an oil spill is the contact of oil with shorelines and organisms that inhabit the water's surface, including birds, marine mammals and reptiles. Birds are highly vulnerable because they spend most of their time on water (diving birds) or shorelines (wading birds). Birds and fur-bearing animals, such as seals, sea otters and sea lions, are immediately affected because oil sticks so readily to feathers and fur. When waterfowl and fur-bearing animals come in contact with oil they usually succumb by loss of buoyancy and hypothermia because oil destroys the insulating properties of plumage and fur. Smooth-skinned mammals, such as dolphins and whales, are generally less affected by direct contact with oil. Ingestion of oil, often from grooming to remove oil from fur or feathers, causes anemia, pneumonia, intestinal irritation, kidney damage, altered blood chemistry, decreased growth and decreased production and viability of eggs. Among reptiles, turtles are considered the most susceptible to surface

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oiling, oiling of nests on beaches or direct ingestion of oiled prey. Possible effects of oil on turtles can include egg and hatchling mortality, a reduction of hatchling size and weight and an increase in respiration rate (RPI 1991).

Mortality due to hypothermia is not instantaneous. Lightly-oiled animals can sometimes recover by grooming to remove small traces of oil, and many animals can be rescued and cleaned. Rates of recovery will depend on: the extent of oiling; the capacity of different species to tolerate the stresses of capture, cleaning, recovery in captivity and re-introduction to the wild; the dose of oil consumed by grooming before rescue; and the size and technical capacity of the rescue effort (Clumpner 2015). Clumpner (2015) identified a need for professional organizations to quickly rescue and treat affected animals.

The littoral zones of aquatic ecosystems extend from the high water mark of shorelines to relatively shallow waters along shore, including the intertidal areas of marine ecosystems. The littoral zone is quite sensitive to stranded oil because it serves as permanent and seasonal habitat for many aquatic species uniquely adapted to periodic exposure to air (e.g., during tidal cycles), turbulence and a wide range of temperatures. Oil may smother sessile species of invertebrates or plants that inhabit the littoral zone when winds, tides or currents bring oil ashore. If the coating of oil is sufficiently thick, it prevents access to sunlight, oxygen, soluble plant nutrients (e.g., N, P), planktonic organisms for filter feeders and removal of waste by flushing with water. It will also impede the movements of organisms to feed, reproduce and avoid predation. This is especially important in low energy environments, where layers of oil are not removed by wave action. For salt marsh plants, oiling of the lower portion of plants and roots is more damaging than coating of leaves and stems, especially if oiling occurs outside the growing season. More damage is experienced if there is repeated contamination of sediments in areas where the oil may persist. Shorebirds and wading birds can be affected by oil in the intertidal area. Land animals, such as raccoons, that scavenge for food in intertidal areas and use them for shelter may ingest oil while eating exposed prey and may become coated in oil while exploring exposed flats and grass beds. The challenge for shoreline cleanup is to remove the oil without killing the organisms that are coated. In most cases, this is impossible, and often the best that can be done is to remove the oil quickly and completely to allow re-colonization of the affected area by healthy organisms from adjacent un-oiled habitats.

4.1.2 Plants

Lewis and Pryor (2013) provided a comprehensive review of publications on the toxicity of 41 crude oils and 56 dispersants to 85 species of unicellular and multicellular algae, 28 wetland plants, 13 mangroves and nine seagrasses. *However, the authors concluded that the database was inadequate to support ERAs of oil toxicity to algae and plants, primarily because the high diversity of methods and measurements (e.g., 107 different response parameters) prevented generalizations*. Effect concentrations varied by six orders of magnitude (0.002 to 10,000 mg/L) and the most sensitive response was embryo fertilization of *Fucus*. Few useful endpoints were derived for risk assessments because most studies were in response to large marine spills and were experimentally diverse. For dispersants, most studies were of formulations that are no longer used, although there have been a few studies of Corexit® EC9500A and Corexit® EC9527A. For example, EC9527A was about 10-40 times less toxic than EC9500A to germination rates of the brown macroalga, *Phyllospora comosa.*

Although oil toxicity to phytoplankton species has been reported frequently (e.g., Garr et al. 2014), it has been difficult to determine if major oil spills cause irreversible damage. Effects on plankton are often mitigated by the rapid dispersion and weathering of oil, recruitment of unexposed organisms from adjacent areas due to currents and mixing, and rapid growth rates.

The primary interactions between oil and phytoplankton may be the physical adherence of oil onto their surface (Lee et al. 1985) and the capacity of algae to absorb hydrocarbons and mediate oil transport within the ecosystem as a result of filter feeding and/or sedimentation processes (Beyer et al. 2015).

Shoreline and riparian plants are another matter. An experimental application of weathered medium-light MESA (Medium South American crude oil) to a Nova Scotia salt marsh suppressed the growth of the predominant plant species, *Spartina alterniflora* (Lee et al. 2003a). Although natural recovery occurred within the same growth season, there was a secondary longer-term change in the plant community; *Salicornia*, a more tolerant and opportunistic plant species, increased in relative abundance. Oil toxicity to salt marsh plants also occurred following the DWH oil spill (Beyer et al. 2015). For some species, only the emergent portion of the plant was affected, and vegetative cover was restored following cleanup. However, where oil penetrated sediments, the roots died and marsh and coastal shorelines were subject to accelerated erosion. This effect was halted only when plants recovered and re-invaded several years after initial oiling. *The role of chemical dispersants in facilitating the transport of oil into marsh sediments is unknown (Lewis and Pryor 2013).* Following the EVOS, aggressive cleaning of shorelines caused significant impacts on abundance and reproduction of the marine macrophyte, *Fucus sp.,* even though it was relatively insensitive to oil (Van Tamelin and Stekoll 1996). The loss of this dominant plant caused a major reduction in habitat and food for associated populations of coastal invertebrates and fish. Full recovery of *Fucus* production required four years or more in the most affected areas.

The effects of oil on freshwater plants are much less studied (Lewis and Pryor 2013), although oil tolerance has been observed in species of bulrushes. Transplants of the three-square bulrush sedge (*Scirpus pungens*) collected from the shores of the St. Lawrence River survived, grew and produced new shoots in sediments contaminated with MESA over a range of concentrations comparable to those associated with oil spills (Longpré et al., 2000). Similarly, a spill of a heavy fuel (HFO 7102) into Wabamun Lake, AB, had little direct effect on the abundance and productivity of softstem bulrush, *Schoenoplectus tabernaemontani* (Thormann and Bayley 2008; Wernick et al. 2009). About 149,500 L entered the lake after accumulating particles of soil and vegetation (Debruyn et al. 2007; Hollebone et al. 2011). Most remained close to shore, contaminating reed beds and sinking to the sediments as tar balls (Hollebone et al. 2011). Bulrush biomass, cover, height and seed head production were unaffected despite heavy oiling and residual contamination of sediments. Nevertheless, there was a major impact on bulrushes caused by the cutting of oiled bulrushes at the sediment surface and the flushing and/or vacuuming of contaminated sediments (Thormann and Bayley 2008). The amount of oiled plants was reduced successfully, but so was the density and biomass of plant rhizomes. The severity of impacts increased with water depth and included lower rates of seed germination, seedling growth, rhizome survival and re-growth. As the predominant shoreline plant, the bulrush provided habitat for nesting aquatic birds, including western grebes (*Aechmophorus occidentalis*), and reproduction of forage fish species and northern pike (*Esox lucius*). Overall, the physical damage to bulrushes of cleanup operations severely reduced the quality of habitat for key aquatic species for several years. Residual tar balls, some of which leaked fresh oil, also persisted for up to 18 months (Hollebone et al. 2011).

Following the June 2012 spill of light sour crude to the Red Deer River, AB, Rood and Hillman (2013) assessed the response of vegetation to floodplain and shoreline oiling and subsequent natural recovery at a previously-established field site. Over the summer period, young balsam poplar trees lost leaves that had been oiled, and growth was checked. However, as the oil weathered and dried on the trees, new shoots emerged on contaminated stems and from roots underneath contaminated surface soils. The overall, effects on trees were limited and due primarily to oil coatings that blocked carbon dioxide uptake and water transpiration. This study demonstrated that leaving oiled vegetation undisturbed to recover on its own prevented habitat damage due to cleanup and avoided colonization of damaged riparian lands by invasive plants. However, the report was limited to one field season only, and no data were presented on impacts on other types of vegetation or animals typical of the riparian zone.

In northern freshwater ecosystems, Hellebust et al. (1975) demonstrated that experimental spills of Norman Wells crude oil to a lake in the MacKenzie River valley caused little effect on planktonic algae in open water other than a blue-green algal bloom. However, the growth of attached filamentous algae was

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inhibited in open waters, as was the growth and developme nt of sedge species (emergent vegetation) in wetlands at the edge of the lake and in oiled plots back from the edge of the lake. Effects were characterized by the death of roots and emergent tissue of *Carex* species, and a lack of chlorophyll and growth of *Equisetum sp* (horse tails) and moss (*Scorpidium scorpiodes*). Initial effects were caused by the thick layer of oil that limited light penetration, but later effects were associated with the redistribution of oil to the root zone of macrophytes and the accu mulation of oil on substrates that supported periphyton. *Wetland macrophytes were highly sensitive to oil, particularly at the margins of open water, with implications for overall productivity of wetlands and the destruction of habitat for aquatic or semi - aquatic animal species.*

<u>Recommendation:</u> Toxicity tests with plants should be conducted to define clear exposure-response relationships, median effective (EC50) and threshold concentrations, and tissue concentrations associated with effects. Field studies are needed to determine the comparative fate, behaviour and effects of oil in wetlands with varied hydraulic regimes (bogs, fens, marshes, swamps), including areas of permafrost. Habitat restoration methods for shoreline vegetation need to be developed to ensure the rapid recovery of shoreline ecosystems and the optimum balance between oil cleanup and habitat protection.

4.1.3 Invertebrates

The toxicity of crude and refined oils to invertebrates is derived directly from laboratory toxicity testing, and indirectly from field studies of individual species or changes in structure of exposed invertebrate communities (usually benthic invertebrates because they are relatively immobile and provide spatial patterns of response). There is a reasonable database of acute med ian lethal concentrations (LC50) for freshwater and marine species, but variations in sensitivity among species with different habitats, life histories and developmental stages (i.e., embryos vs. adults) are not fully understood. For example, Lee et al. (2 000) found differences in sensitivity between the mystery snail (*Viviparus georgianus*) and the mimic pondsnail (*Pseudosuccinea columella*) in a controlled oil spill experiment at a wetland site along the St. Lawrence River (Ste. Croix, QC) that were attribu ted to feeding habits. *V. georgianus* , a detritivore, assimilated contaminants directly from sediments, while *P. columella*, a herbivore, assimilated contaminants indirectly, presumably from oiled vegetation. Although laboratory tests may provide an objective and precise measure of toxicity, toxicity is not absolute and varies with species, life stage and test conditions (Section 4.3), as well as the properties of the oil tested. *For acute lethality, species sensitivity distributions implied that marine invertebrates were no more or less sensitive to a variety of test oils than fish species* (Barron et al. 2013; details in section 4.2.1). As observed for other aquatic species, dissolved petroleum hydrocarbons appear more bioavailable to invertebrates than particulate or droplet oil (reviewed by Dupuis and Ucan-Marin 2015).

Effects on invertebrate species and communities were evident following the EVOS. The dominant taxa in intertidal communities affected by the spilled oil and cleanup procedures included: limpets, barnacles (three species), mussels, snails (two species) and oligochaetes (worms) (Highsmith et al. 1996; Hooten and Highsmith 1996). Effects were caused by oil toxicity, smothering and the loss of habitat when *Fucus* beds were destroyed by cleanup activities. Cleanup procedures, such as washing, steaming or mechanical removal, destroyed some animals , such as grazers (e.g., periwinkles), or their food supply (biofilms on rocks). Two groups, barnacles and oligochaetes, increased in abundan ce because they rapidly colonized space left by missing species and they consumed oil-degrading microbes. Potential impacts on other organisms would include a reduction in food supply for consumers of intertidal plants and invertebrates, such as shorebirds, diving ducks, otters and even bears. Recovery was incomplete two years following the spill and contamination of mussel beds, including the mussels themselves, persisted more than three years after the spill (Babcock et al. 1996).

Following the DWH spill, laboratory experiments demonstrated the acute toxicity of mechanically - and chemically-dispersed MC -252 (Macondo) oil to crabs (*Rhithropanopeus harrisii*, *Callinectes sapidus*) (Anderson et al. 2014) and mysid shrimp (*Americamysis bahia*) (Hemmer et al. 2011). In contrast, field studies during the spill found surprisingly few effects on invertebrates, with no reports of mass mortalities on oiled shorelines or in coastal waters (reviewed by Beyer et al. 2015). Growth rates of shrimp were reduced in coastal emb ayments affected by the spill (Rozas et al. 2014) , likely due to the closure of shrimp fisheries which increased their abundance (van der Ham and de Mutsart 2014) and depleted their food supply (Beyer et al. 2015). Although caged filter-feeding mussels and oysters were exposed to oil - contaminated particulates during the spill, there was no evidence of assimilation of oil -derived carbon in samples collected after the spill, as indicated by stable isotope ratios in shell and soft tissues (Carmichael et al. 2012b; Fry and Anderson 2014).

Oil dispersants enhanced the exposure, uptake and toxicity of oil to invertebrates, but the opposite was true for fiddler crabs (*Uca minax*) perhaps because of complex relationships among oil droplet si ze, methods of dispersion and species -specific pharmacokinetics (Chase et al. 2013) . For example, in a laboratory exposure of the copepod *Calanus finnmarchicus* to oil droplets from physically - and chemically-dispersed Troll oil, filtration rates were inver sely proportional to oil concentrations (Nordtug et al. 2015), due to avoidance, clogging of copepod filtering structures or toxicity. Nevertheless, the filtering rates were sufficiently high that zooplankton could conceivably enhance the rate of removal o f oil from water (Nordtug et al. 2015).

As with fish species, invertebrate embryo development is particularly sensitive to oil exposure. Exposure of adult sea -ice amphipods (*Gammarus wilkitzkii*) to a continuous flow of water from an oiled bead column caused little effect on adults but led to a high frequency of abnormalities in embryos carried in their brood pouch (Camus and Olsen 2008). The fertilization success of oyster eggs (*Crassostrea virginica*) was diminished by oil exposure, as were trocophore and D-stage development, with subsequent increased rates of deformities, reduced overall survival and reduced activity of survivors (Laramore et al. 2014). These data suggest that trends and conclusions from studies with fish embryos (Section 4.1.4.2) may also apply to invertebrate embryos.

The potential impacts of oil spills on terrestrial invertebrate species that interact with emergent vegetation in salt marshes are not well studied (Beyer et al. 2015) , a research need also relevant to freshwater riparian lands affected by oil spills (e.g., the Kalamazoo River spill, Chapter 8).

<u>Recommendation:</u> Research is needed on interactions between the physiology, life history and habitat characteristics of invertebrates and susceptibility to oil exposure and toxicity.

4.1.4 Fish

The toxicity of oil to fish is treated in greater detail than for other aquatic species, primarily because more research has been done. Many of the interactions between the chemistry of oil and its toxicity to fish may also apply to other species.

4.1.4.1 Acute toxicity

Fish are at risk of acute exposure to oil in the 24 to 48 -hour period following a discrete spill to large marine or freshwater ecosystems. However, fish kills are typically brief and localized because of the rapid loss of the acutely lethal low molecular weight (LMW) components of oil due to dilution and weathering. In contrast, acute mortality can be extensive when there is an ongoing point source of oil, e.g., in a high -gradient river where oil is distributed rapidly but not highly diluted bef ore weathering occurs (e.g., pipeline break, Pine River, BC, reviewed in Hodson et al. 2011).

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The acute lethality of oil is associated primarily with LMW components, such as monoaromatics (e.g., BTEX), diaromatics (naphthalene and alkyl naphthalenes) and short-chain alkanes (< C12) (NRC 2005). Metals, such as aluminum, copper, nickel, vanadium and mercury, occur in most crude oils, with concentrations increasing from light to heavier crudes (reviewed by Dupuis and Ucan-Marin 2015). However, these metals have not been associated with the acute toxicity of oil to aquatic biota, either because concentrations are sub-toxic, or the metals are not sufficiently bioavailable in acute exposures. Environmental contamination by metals, particularly mercury and vanadium, is evident near oil production facilities (Lee 1983; Kelly et al. 2010). The sources are predominantly extraction processes, and in the case of the Alberta oil sands, airborne and waterborne particulates released by the erosion of surface soils during bitumen mining.

Sublethal effects of acute exposure are also evident and include biochemical signs of exposure to polycyclic aromatic hydrocarbons (PAHs) (e.g., Ramachandran et al. 2004a), avoidance of oil solutions and suppression of feeding behaviour (Lari et al. 2015) and delayed effects of embryo toxicity, such as impaired swimming of juvenile fish (e.g. Mager et al. 2014; Section 4.7.1.2). However, most publications on acute exposures concern only the lethality of petroleum hydrocarbons and modeling for risk and impact assessments.

The acute lethality of oil is usually attributed to LMW hydrophobic petroleum hydrocarbons that partition into lipid membranes and cause mortality by narcosis, a general term for an array of effects on lipid membrane receptors and functions (Campagna et al. 2003). Narcotics include non-polar compounds (e.g., benzene, naphthalene) and polar compounds (e.g., heterocyclic hydrocarbons; Table 2.1), which differ somewhat in their water solubility, bioavailability and toxicity. Larger compounds, such as alkyl PAHs (alkyl PAHs) with three or more rings, contribute less to acute toxicity because they are taken up more slowly than LMW compounds (McGrath and Di Toro 2009). Narcotics are sufficiently water soluble to reach maximum concentrations in water in contact with fresh, un-weathered oil within two hours (NRC 2005), but *most are lost from water within two days by volatilization, biodegradation and dilution (Chapter 2). As a consequence, lethal concentrations in water are not sustained unless there is an ongoing discharge of oil (NRC 2005).*

The acute toxicity of hydrocarbons to aquatic biota can be estimated from octanol-water partition coefficients (K_{ow}) (McCarty and Mackay 1993). Aquatic organisms exposed to solutions of hydrocarbons accumulate them in proportion to K_{ow} **(Figure 4.1)**. Mortality occurs when tissue concentrations reach a 'critical body residue' or median lethal dose (CBR or LD50) of approximately 5 mM in tissue lipid (McCarty and Mackay 1993). There are linear relationships between K_{ow} and BCFs, or LD50s across an array of hydrocarbons (**Figure 4.1**), so that LC50s for individual hydrocarbons can be estimated mathematically from K_{ow}. Curvilinearity reflects reduced rates of chemical uptake due to the increased miscibility of compounds with water at lower values of K_{ow}, and steric interference of diffusion of larger compounds through membranes at higher values of K_{ow}. Within the linear range of K_{ow} models, mixtures of hydrocarbons appear to act additively when expressed as toxic units (TU), allowing the estimation of the toxicity of mixtures from the partial TU contributions of each component of the mixture.

> **K_{ow} (octanol-water partition coefficient)**: the ratio of the concentrations of a compound in *n*-octanol (a water-insoluble phase that mimics tissue lipids and floats on water) to the concentrations in underlying water, at equilibrium.
>
> **BCF** - The bioconcentration factor or ratio of concentrations in tissue to concentrations in water.

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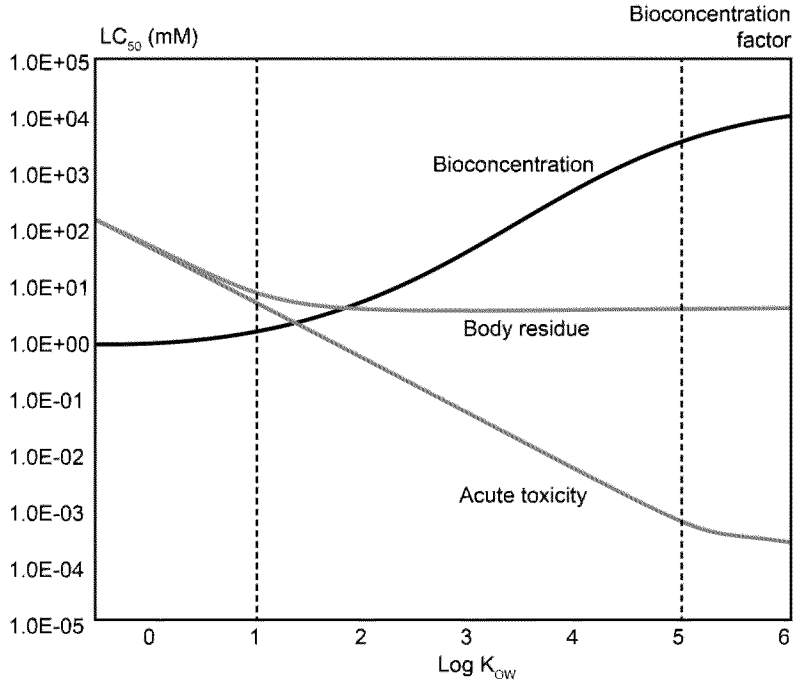

Figure 4.1. *The general relationships among the octanol -water partition coefficients (K_{ow}) of neutral organic compounds and their bioaccumulation and acute toxicity (narcosis) to fish. The horizontal portion of the 'body residue' regression represents the critical body residue or median lethal dose (LD50) of 5 mM common to most narcotics. Adapted with permission from McCarty, L.S., and D. Mackay. 1993. Enhancing ecotoxicological modeling and assessment. Body residues and modes of toxic action. Environmental Science &.Technology 27:1718 - 1728. Copyright (1993) American Chemical Society.*

These predictable relationships among properties of hydrocarbons form the basis for mathematical models to estimate the total dose received, the dose-rate and the acute lethality of mixtures of hydrocarbons from the concentrations measured in water. These include the target lipid model (TLM) (Di Toro et al. 2000; McGrath and Di Toro 2009); the Oil Toxicity and Exposure model (OILTOXEX)(French-McCay 2002); the PETROTOX model (Redman et al. 201 4); and ecological models for the Barents Sea (Olsen et al. 2013). These models provide a very convenient tool for estimating exposure of aquatic organisms to hydrocarbons, and the risk of acute lethality following oil spills . However, they are less successful in predicting sublethal toxicity associated with highly specific mechanisms of action (MOAs) such as interactions of PAHs with specific cardiac receptors (Section 4.1.4.2).

Toxic Unit (TU): The ratio between the concentration of a compound in water and its LC50. When water concentrations exceed the LC50 (> 1.0 TU), test organisms will die rapidly; when less than the LC50 (<1.0 TU), mortality occurs more slowly and fewer organisms die.
- Two compounds act additively when 0.5 TU of compound A plus 0.5 TU of compound B (or 0.3 of A and 0.7 of B) cause 50% mortality of exposed organisms.
- Two compounds are termed synergists when toxicity appears more-than-additive.
- Two compounds are termed antagonists when toxicity appears less-than-additive.

A primary obstacle to modeling the acute lethality of actual spills is often the inadequate characterization of the toxic components of oil or water samples (NRC 2005). The models are also based on a variety of assumptions (text box) and cannot be validated before spills. Validation following a spill is challenging due to the problem of finding dead organisms and characterizing their exposure to oil. Variance about regressions relating log LC50 to log K_{ow} is also large, with predicted values often one - to two-orders of

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magnitude above or below measured values (McGrath and Di Toro 2009), and model errors are rarely incorporated into quantitative uncertainty estimates in ecological risk assessments. The original LC50 values from which the models were first derived (e.g., Di Toro et al. 2000; French-McCay 2002) were also generated prior to 2000 from a heterogeneous array of chlorinated and non-chlorinated compounds and may not be representative of all petroleum hydrocarbons, especially for compounds with K_{ow} values greater than 5.3 (See section 4.3).

Assumptions inherent in models of oil mixture toxicity based on K_{ow} and narcosis

- All chemicals in the TLM model share a single common mode of action (MoA) of narcosis;
- Hydrocarbon concentrations in water are constant with time;
- Tissue lipid concentrations are at equilibrium with water concentrations;
- Metabolism of hydrocarbons does not change tissue concentrations or rate of exposure;
- Metabolites of hydrocarbons do not differ in their toxicity from parent compounds;
- Concentrations of compounds in mixtures are adequately characterized;
- The K_{ow} values of chemicals used to develop models encompass the range of K_{ow} values for petroleum hydrocarbons that contribute to acute lethality of oil;
- The models can predict the toxicity of hydrocarbons with log $K_{ow} > 6$, beyond the linear range of the statistical relationship (0-5.3) (McCarty and Mackay 2003);
- Models developed with data from un-substituted PAHs apply to alkyl PAHs;
- There are no synergistic or antagonistic interactions in mixtures, only additivity;
- There are no mixture interactions with other compounds in water, natural or anthropogenic; and
- Only dissolved hydrocarbons are present in the aqueous phase; non-aqueous phase oil (droplets) is not present.

Recommendation: Models predicting the acute lethality of oil should be refined to narrow confidence limits about model predictions by restricting toxicity data for model development to petroleum hydrocarbons and to single test species.

4.1.4.2 Chronic and sublethal toxicity

Chronic and sublethal effects of oil on fish are associated with either chronic exposures (e.g., oil stranded in sediments) or with the delayed or lingering effects of acute exposures. Effects range from the first genetic and molecular responses of cells to impacts on rates of reproduction, growth, disease and survival. Exposures to oil may be via respiratory uptake of hydrocarbons from water (most common), direct contact with droplets (an emerging area of research), the diet or maternal transfer to eggs (not well studied). Most studies of waterborne oil reference the concentrations of total PAHs (TPAHs) and/or total petroleum hydrocarbons (TPHs) in test solutions that affect 50% of test organisms. The EC50 and LC50 values for chronic exposures range from 0.3 –60 µg/L TPAH and 0.03 – 11 mg/L TPH (**Figure 4.2, Appendix D, Table D.1**; Hodson et al. 2011; Logan et al. 2015). These concentrations are well within the range of concentrations anticipated in surface waters near slicks of un-dispersed oil or in plumes of chemically-dispersed oil and only brief exposures (one to 96 hours) to these concentrations can cause embryo toxicity (McIntosh et al. 2010; Section 4.2.3). The variance among studies reflects differences in species sensitivity, the mix of toxic components unique to each oil (Section 4.2.2), and the methods of preparing, testing and characterizing solutions of oil (Section 4.3).

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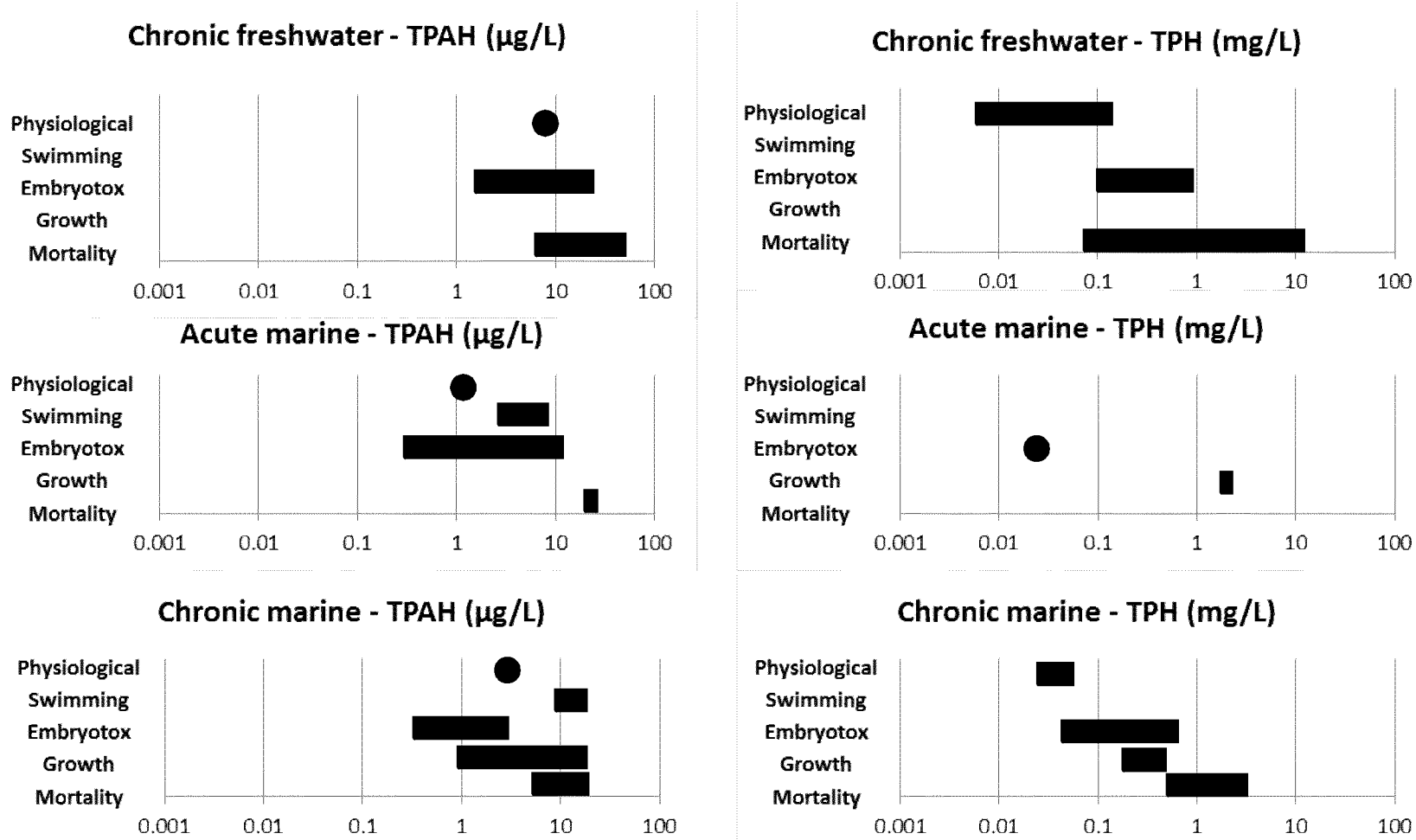

Figure 4.2 *Sublethal (EC50 values) and lethal (LC50 values) concentrations of total polycyclic aromatic hydrocarbons (TPAH) and total petroleum hydrocarbons (TPH) reported for fish subject to acute (1 -96 h) or chronic (>96 h) exposures to oil. The black rectangles represent t he range of concentrations reported to cause a given effect, and the solid circles indicate only one measurement was available. Data were drawn from stu dies reporting measured concentrations (**Appendix D, Table D.1** : Kocan et al. 1996; Marty et al. 1997; Car ls et al. 1999, 2008; Little et al. 2000; Brand et al. 2001; Couillard et al. 2005; Greer et al. 2012; Gulec and Holdway 2000; Pollino and Holdway 2002; Perkins et al. 2005; Shen et al. 2010; Olsvik et a l. 2011; Wu et al. 2012; Gardiner et al. 2013; Incardona et al. 2014; Mager et al. 2014; Martin et al. 2014; Dussauze et al. 2015; Madison et al. 2015; Sørhus et al. 2015).*

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For juvenile or adult fish, the overall effect of oil exposure is to reduce fish production at spill sites (Reviewed by Dupuis and Ucan -Marin 2015). Oil toxicity may include reduced rates of growth and increased rates of infectious disease (Section 4.1.4.4), mutations and possibly cancer (Section 4.1.4.5) and disruption of sexual maturation and reproduction.

Reproductive effects include lower concentrations of sex steroids, such as plasma estradiol, slower gonadal growth and delayed time to sexual maturity (reviewed in Brown-Peterson et al. 2014). These effects are more typical of chronic rather than delayed toxicity, and may involve endocrine disruption by hydrocarbons that mimic steroid hormones, or increased rates of hormone metabolism by enzymes that oxygenate hydrocarbons. Establishing links between oil exposure and effects on reproduction is complex because effects depend on the stage of maturation when fish are first exposed (Truscott et al. 1983). *Identifying and interpreting the effects of oil spills on reproductive performance of fish will be complicated by a high variability in their sensitivity to endocrine disruption—timing is critical!*

Research following the EVOS highlighted the high sensitivity of fish embryos to oil exposure, as judged from the low concentrations of hydrocarbons causing toxicity (e.g., Carls et al. 1999; Appendix D, Table D.1). Ecologically, embryo sensitivity is also imparted by their inability to avoid exposure to hydrocarbons. Embryo-toxicity is often considered an effect of chronic exposure, i.e., an exposure that constitutes a significant portion of an organisms' life span or development stage. However, there are instances of delayed effects, where a response typical of a chronic exposure occurs long after a brief or acute exposure (Section 4.2.4). *Delayed effects may not be obvious at the time of a spill, but may become evident as changes in rates of recruitment, growth, reproduction, disease or mortality in the months or years following a spill, depending on the life history of the species exposed*. In this review, chronic toxicity will refer to effects of exposures that exceed 96 hours or effects of acute exposures that are evident 96 hours or more following exposure.

Embryo toxicity due to exposure to oil or to individual PAH is characterized by blue sac disease, a syndrome that includes: pericardial and yolk sac edema; cardiotoxicity; craniofacial, ocular and spinal deformities; degeneration of muscle somites, peripheral neurons and endothelial cells; fin erosion; failure of swim bladders to inflate; altered behaviour and swimming ability; and induction of cytochrome P-450 (CYP1A) enzymes (e.g., Schein et al. 2009; deSoysa et al. 2012; Incardona et al. 2013; details in Section 4.1.4.4). Not all of these signs of toxicity are evident in embryos of each species exposed to oil, but all species exhibit many of these signs.

The high sensitivity of embryos to oil toxicity is due to the large number of genes that are up-regulated or activated only during embryonic development. It is critically important that these genes be activated in the proper sequence, to the proper extent and at the proper time for normal development. However, PAHs, the components of crude oil associated with embryo toxicity (Section 4.1.4.3), can bind to protein receptors in cells to interfere with protein function or to activate a broad spectrum of genes. Thus, oil exposure interferes with the carefully orchestrated sequence of events essential for normal development of tissues such as the heart, leading to impaired cardiac function (e.g., Incardona et al. 2004, 2009; Hicken et al. 2011).

> **Cytochrome P450:** The cytochrome P450 gene (*cyp1a*) codes for CYP1A proteins, enzymes that oxygenate PAHs as the first step in their excretion. 'CYP1A induction' refers to the increased synthesis of CYP1A proteins when the *cyp1a* gene is up-regulated after specific PAHs or alkyl PAHs bind to the intracellular aryl hydrocarbon receptor protein (Billiard et al. 2002).

The up- and down-regulation of a large number of genes represent the primary reactions to oil exposure, indicating that hydrocarbons have interacted directly with cellular receptors to initiate gene transcription. Activation of one gene can cause the secondary activation of many more, because they are often linked

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directly or indirectly. One of the most commonly reported responses to exposure to crude oil and to chlorinated hydrocarbons such as PCBs, is activation or up-regulation of the *cyp1a* gene, a member of a large family of cytochrome P450 genes (e.g., Olsvik et al. 2011, 2012; Sørhus et al. 2015). There are many genes involved in embryo development that are 'downstream' of *cyp1a* and that are also activated with exposure to PAHs. Activation of these genes at an inappropriate stage in embryonic development may be one cause of PAH and dioxin-like toxicity (Billiard et al. 2008).

The CYP1A proteins are enzymes that oxygenate double bonds in PAHs as the first step in their excretion. Thus, increases in CYP1A enzyme activity following gene activation cause a rapid increase in the rate of excretion of PAHs. For example, the half-life of tissue retene (7-isopropyl-1-methylphananthrene, a C-4 alkylphenanthrene) in rainbow trout is estimated as <14 hours at 15 °C (Fragoso et al. 1999). High rates of oxygenation of PAHs can prevent acute lethality by facilitating the clearance of PAHs from tissues (Hicken et al. 2011). *Not all PAHs will induce CYP1A enzymes (Basu et al. 2001; Billiard et al. 2002), but all PAHs accumulated by fish exposed to oil are subject to oxygenation by CYP1A enzymes because mixtures of PAHs in oil include those that cause induction.* In vertebrate species exposed to PAHs in oil, the up-regulation of *cyp1a* can increase CYP1A enzyme activity by up to 1,000-fold, in parallel with increasing toxicity (Brinkworth et al. 2003; Colavecchia et al. 2007; Madison et al. 2015). Thus, CYP1A induction provides a reliable indicator of PAH exposure and potential effects in vertebrates; invertebrate species do not express *cyp1a* genes and excrete PAHs more slowly than vertebrates.

The oxygenation of PAHs can also release reactive oxygen species (ROS) and reactive metabolites of PAHs into the cell (reviewed by Crowe et al. 2014). There are frequent reports that genes involved in combating oxidative stress, suppressing mutations associated with the effects of ROS on DNA, and responding to general cellular stress are up-regulated in oil-exposed fish embryos (e.g., Olsvik et al. 2011; Madison et al. 2015). Exposure of cod larvae (*Gadus morhua*) to mechanically-dispersed oil caused the expression of genes related to drug metabolism, endocrine system development and function and lipid metabolism, with the highest response for *cyp1a* (Olsvik et al. 2011). Genes related to bone resorption were up-regulated while those related to bone formation were down-regulated, consistent with craniofacial and spinal deformities frequently reported in oil-exposed embryos.

Unlike acute lethality, there are multiple MOA that contribute to embryo toxicity, but they are not mutually exclusive. Craniofacial deformities reflect the disruption of the normal transformation of neural crest cells to specific cell and tissue types (de Soysa et al. 2012). Cardiac toxicity represents unique interactions of PAHs with cardiac receptors, so that the heart does not develop to its proper form and function (Incardona et al. 2004, 2006, 2013; Hicken et al. 2011; Scott et al. 2011). These interactions were evident *in vitro* as impaired ion regulation, cell signaling and contractile rhythmicity in oil-exposed cardio-myocytes of blue-fin (*Thunnus thynnus*) and yellow-fin (*Thunnus albacores*) tuna (Brette et al. 2014). Cellular effects corresponded to changes in cardiac function in whole embryos when oil-exposure impaired atrial-ventricular signaling and coordination (Incardona et al. 2009). Edema and circulatory failure of embryos may follow impaired cardiac function or be due to oxidative damage to lipid membranes in endothelial cells of blood vessels following CYP1A metabolism of PAHs. CYP1A induction was evident early in embryonic development and was prominent in gills, liver, kidney, eyes and endothelial cells of blood vessels of fish embryos exposed to retene (Brinkworth et al. 2003), bituminous sediments (Colavecchia et al. 2007) or oil (Incardona et al. 2013).

The predominant MOA underlying chronic toxicity varies with the PAH being tested. For example, the embryo toxicity of pyrene depends on its interaction with the arylhydrocarbon receptor protein and CYP1A induction (Incardona et al. 2006). In contrast, phenanthrene, retene and dibenzothiophene are toxic even when CYP1A protein synthesis is blocked (Incardona et al. 2004, 2005; Scott et al. 2011). In fathead minnow and white sucker embryos exposed to bituminous sediments, CYP1A enzymes were induced in every tissue showing histopathology, and the extent of induction was correlated to rates of

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mortality and pathology (Colavecchia et al. 2007). Although correlations imply a direct role of CYP1A enzyme activity in toxic effects, they may simply reflect PAH concentrations in each tissue.

The metabolism of PAHs can also increase their toxicity and carcinogenicity (Section 4.1.4.5). Inhibition of CYP1A induction demonstrated that metabolites of retene appear more toxic to trout embryos than the un-metabolized parent compound (Hodson et al. 2007b; Scott and Hodson 2008; Scott et al. 2009). Similarly, phenolic derivatives of 1-methylphenanthrene were up to four-fold more toxic to Japanese medaka (*Oryzia latipes*) embryos than un-substituted 1-methylphenanthrene (Fallahtafti et al. 2012). The pattern of –OH substitution affected toxicity, and may reflect the propensity for biotransformation of phenolic derivatives to *p*-quinones. Unlike acute lethality, chronic toxicity was unrelated to solubility or K_{ow} because all four OH-substituted congeners had the same modeled K_{ow}. Similarly, co-exposure of trout embryos to phenanthrene (does not induce CYP1A) and to an inducer of CYP1A enzymes caused greater toxicity to trout embryos than exposure to phenanthrene alone (Hawkins et al. 2002), consistent with the idea that metabolites were the toxic forms of PAHs. The MOA of alkyl PAHs likely includes oxidative stress because the severity of embryo toxicity and signs of oxidative stress were reduced when rainbow trout (*Oncorhynchus mykiss*) embryos were co-exposed to retene and Vitamin E, an antioxidant (Bauder et al. 2005). *The balance between the accelerated rates of excretion of PAHs and the production of excess reactive intermediates will determine whether oxidative stress and pathology occur and will depend, in turn, on the extent and duration of oil exposure and accumulation of TPAHs.*

Research and monitoring of the effects on fish of the DWH spill focused largely on embryo toxicity in laboratory or mesocosm experiments. The MC-252 oil caused similar cardiotoxicity as Alaska North Slope (ANS) crude oil, and pathology and up-regulation of *cyp1a* genes by both oils were correlated to concentrations of TPAH (Incardona et al. 2013). Oiled sediments from affected coastal embayments caused the same toxic effects on zebrafish embryos as reference sediments spiked with MC-252 oil (Raimondo et al. 2014) and were consistent with effects caused by chemically-dispersed oil, with a threshold sediment concentration of TPAH between 2 to 27 mg/kg. Gulf killifish (*Fundulus grandis*) embryos exposed to sediments from coastal areas contaminated by the DWH oil spill responded with the typical signs of embryo toxicity (Dubansky et al. 2013) Similarly, killifish from areas contaminated by oil showed the down-regulation of 1,070 genes and the up-regulation of 1,251 relative to un-oiled reference areas (Garcia et al. 2012) and many gene responses indicated physiological and reproductive impairment (Whitehead et al. 2012). The genomic responses were reproducible in laboratory exposures of killifish (Pilcher et al. 2014) and spotted sea-trout (*Cynoscion nebulosus*) (Brewton et al. 2013) to MC-252 oil, with clear activation of excretory pathways (e.g., *cyp1a*). Activation of *cyp1a* was associated with growth depression in sea-trout (Brewton et al. 2013) and with gene expression in killifish indicative of changes in gene transcription, cell cycle progression, RNA processing, DNA damage, oxidative stress and apoptosis (Crowe et al. 2014; Pilcher et al. 2014). Impacts on the reproductive biology of spotted sea-trout sampled from oil-contaminated coastal embayments of Louisiana and Mississippi were reported by Brown-Peterson et al. (2014). Impacts included delayed maturation, smaller relative gonad sizes and a reduced frequency of spawning, but no biological or chemical indicators of exposure to oil were reported.

In general, there is a good understanding of the waterborne concentrations of TPHs, TPAHs and specific un-substituted and alkyl-substituted PAHs causing chronic toxicity, and sufficient data to support risk assessments of oil spills.

<u>Recommendation:</u> Additional research is needed to reduce the uncertainties in translating laboratory data for risk and impact assessments. In particular, relationships need to be established among waterborne concentrations of hydrocarbons, tissue concentrations, genomic responses and toxic effects. Furthermore, mechanisms of action and critical exposure periods for impacts on the reproductive biology of sexually maturing fish need to determined, as does the role of PAH metabolism in toxicity of oil to fish.

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4.1.4.3 Hydrocarbons causing chronic toxicity

To establish the connection between exposure and environmental impacts of spilled oil, it is essential to know which constituents of oil cause toxicity and at what concentrations. *The components of oil that cause chronic toxicity to fish embryos are considered to be alkyl PAHs (Table 2.1), including heterocyclics such as alkyl dibenzothiophenes* (Rhodes et al. 2005). TPAHs comprise about 0.2 to 6.5% by weight of oil, depending on its source and type, and alkyl PAHs comprise 80-95% of TPAHs in crude and refined oils (Wang et al. 2003). *Alkyl PAHs partition from oil to water more slowly than monoaromatics, reaching peak concentrations within four to 12 hours (NRC 2005). Because of their lower volatility, alkyl PAHs persist in water longer than monoaromatics and are dissipated primarily by advection and diffusion. Over weeks or months, alkyl PAHs stranded on substrates will continue to partition from oil into water* (NRC 2005), as was evident from a comparison of the cardiotoxicity to zebrafish (*Danio rerio*) embryos of two crude oils (Jung et al. 2013).

A number of indirect approaches have been used to identify the components of oil causing chronic toxicity to fish embryos. Correlations between concentrations of alkyl PAHs in test solutions and the chronic toxicity of ANS crude oil (**Table 2.2**, Appendix B) to fish embryo development were reported by Carls et al. (1999). Using a toxic units approach, Barron et al. (2004) evaluated four mechanistic models of the chronic toxicity of the constituents of oil test solutions to pink salmon (*Oncorhynchus gorbuscha*) and Pacific herring (*Clupea pallasi*) embryos and concluded that toxicity was best described by summing the toxic unit contributions of alkyl phenanthrene concentrations.

More direct evidence was provided by studies of the chronic embryo toxicity of individual PAH congeners that produced the same syndrome of blue sac disease as did exposure to crude and refined oils. The PAHs tested are commonly found in oil and included 3- to 5-ringed un-substituted PAH (Hawkins et al. 2002; Incardona et al. 2004, 2006, 2011; Rhodes et al. 2005), alkyl phenanthrenes (Billiard et al. 1999; Kiparissis et al. 2003; Turcotte et al. 2011), hydroxylated derivatives of alkyl phenanthrenes (Fallahtafti et al. 2012), alkyl chrysenes and alkyl benzo[a]anthracenes (Lin et al. 2015), and dibenzothiophene and dimethyl dibenzothiophene (Rhodes et al. 2005). Effects-driven-chemical-fractionation (EDCF) of crude oil (Hodson et al. 2007a) and heavy fuel oil (Adams et al. 2014b; Bornstein et al. 2014) linked embryo toxicity to fractions and sub-fractions of oil that contained 3- to 5-ringed alkyl PAHs. Alkyl PAHs were progressively enriched in serial fractions of HFO in parallel with increased toxicity, and comprised up to 25% of total hydrocarbons in toxic fractions. Fractions dominated by LMW compounds, including BTEX and alkyl naphthalenes (2-ringed PAH), or by high molecular weight (HMW) aliphatics (waxes), resins or asphaltenes (Table 2.1) were non-toxic at the concentrations tested.

Overall, many of the chronic effects of oil on fish embryos can be attributed to 3- to 5-ringed alkyl PAHs. The chronic toxicity of PAHs to fish has been associated with their binding to the AhR protein (Section 4.1.4.2). However, other components of oil may also be AhR agonists, as indicated by *in vitro* reporter gene assays of hydrocarbons extracted from aquatic worms exposed to SARA fractions (Section 2.1.1) of eight different crude and fuel oils (Vrabie et al. 2012). The AhR agonists were aromatic, hydrophobic, resin-like compounds primarily in mid- to high-boiling point fractions, and some were resistant to biotransformation *in vitro*. There were too many compounds in isolated fractions to identify specific agonists, although these characteristics correspond to some un-substituted and alkylated PAHs and heterocyclic aromatic compounds.

In all EDCF studies, most components of toxic fractions were unidentified and may have included polar compounds not typically analyzed in detail in oil. Polar compounds in a biodegraded Norwegian Sea crude oil and in 14 fractions of WAF solutions created by high performance liquid chromatography (HPLC) separation were identified as agents affecting rainbow trout liver cells in culture (Melbye et al. 2009). Responses included induction of CYP1A and vitellogenin (egg yolk protein, an indicator of estrogen-like compounds), as well as cytotoxicity. The main contributors to all three responses were the

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water-soluble components of the polar fraction that contained 70% of the hydrocarbons, while fractions enriched with PAHs and alkyl PAHs were less potent. However, ketone and quinone derivatives of a wide array of 3 - to 6 -ringed non -alkyl PAH s showed significant potency for activating the *cyp1a* gene in mammalian cell assays (Misaki et al. 2007). In contrast, a survey of the endocrine -disrupting potential of 11 crude and refined oils using *in vitro* mammalian cell assays demonstrated that all contained compounds with some estrogenic or androgenic potency, but potencies were 10^4 to 10^7-fold lower than those of the reference compounds estradiol and testosterone (Vrabie et al. 2010, 2011). Although polar compounds, such as phenols, are typically associated with endocrine effects, no equivalent fractionation studies have been conducted to identify whether components of oil cause endocrine disruption *in vivo*.

Similarly, the toxicological significance of the oxygenated hydrocarbons measured in MC -252 oil (Table 2.2) over an 18-month period of weathering following the DWH oil spill is unknown (Aeppli et al. 2012). However, aromatic and polar compounds extracted from biodegraded ANS crude oil were chronically toxic to embryos of inland silversides (*Menidia beryllina*) (Middaugh et al. 2002). Extracts of the saturate fraction were non-toxic at the concentrations tested, but the fractions recombined represented the additive effects of the aromatic and polar compound s. This experiment did not analyze oxygenated compounds or the toxicity of untreated oil, but the y demonstrate the presence of significant concentrations of residual toxic compounds in biodegraded oil. It is essential that oil toxicity tests include as complete an analysis of test solutions as possible, including polar fractions. Routine analyses by GC -MS identify about 40 -50 different PAHs or families of alkyl PAHs, in addition to BTEX and saturates.

<u>Recommendation:</u> Research is needed to identify petroleum hydrocarbons associated with different toxic effects.

Models have been developed for ERAs to predict the chronic toxicity of oil (e.g., TLM) (McGrath and Di Toro 2009). However, chronic toxicity is not modeled directly. Instead, acute lethality (LC50) is calculated by the model, and chronic toxicity is estimated from the average ratio of concentrations causing acute and chronic toxicity (acute -chronic ratio or ACR) for algae, invertebrates and fish. However, observed ACR values ranged 10-fold higher and lower than the average, likely because of large differences in the physiology and biochemistry among widely differing plants and animals used as test species. *All of the assumptions underpinning the acute lethality model (Section 4.1.4.1) are inherent in the chronic toxicity model, with additional critical assumptions about chronic toxicity that weaken its application to risk assessment (text box). Not surprisingly, the TLM model does not predict the chronic toxicity of petroleum hydrocarbons very accurately, and a further 'correction factor' of 2.0 must be used to compensate for the average under-estimate of toxicity, which in itself is variable* (McGrath and Di Toro 2009). While these models have been applied to the results of published toxicity tests with a variety of species and a variety of chemical mixtures (McGrath and Di Toro 2009), the results were not very satisfactory, primarily because the case studies examined were not designed for this purpose.

Assumptions inherent in models of acute lethality applied to chronic toxicity

- The MOAs for chronic toxicity and acute lethality are the same (narcosis), regardless of a wide array of sublethal effects caused by oil (Section 4.1.4.2) and large differences in the genetics and physiologies among algae, invertebrates and fish; and
- A single mean ACR derived from tests with algae, invertebrates and fish predicts the chronic toxicity of all PAH to fish embryos.

Although current models carry a high r isk that predicted toxic effects could be under - or over-estimated by a wide margin, the advantages of modeling mixture toxicity for ERAs and EIAs are very compelling, considering the cost and time needed to run chronic toxicity tests for every oil that might possibly be

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spilled or every oil sample collected at a spill. Recently, Redman et al. (2014) used the PETROTOX model to estimate the chronic toxicity of HFO and fractions of HFO created for an EDCF using the reported concentrations of PAH s in each fraction. There was a good correspondence between toxicity predicted from the PETROTOX model and toxicity measured directly (Adams et al. 2014b), although more research is needed to identify sources of error in both the model and the EDCF.

Recommendation: Models of chronic toxicity must be developed from results of chronic toxicity tests and not from acute toxicity tests via application factors.

4.1.4.4 Physiological and pathological responses

The molecular responses of fish exposed to oil in laboratory experiments (Section 4.1.4.2) suggest that there should be equivalent responses in fish surviving an oil spill and related changes in physiological functions that could affect the performance of those fish. Following the 2007 oil spill from the *Hebei Spirit* off the coast of Korea, Jung et al. (2011, 2012) measured the activity of liver CYP1A enzymes and the concentrations of PAH metabolites in bile of two benthic species, marbled flounder (*Pseudopleuronectes yokohamae*) and flatfish (*Paralichthys olivaceus*), sampled in 2008 and 2009. These biomarkers[1] provided clear evidence that fish were accumulating and responding to PAH s from the oil spill in a spatial pattern that corresponded to measured concentrations of PAH s in sediments. The consistent responses rela tive to fish from a reference area during the two years following the spill indicated the persistence of bioavailable hydrocarbons in sediments.

Balk et al. (2011) also observed similar CYP1A and bile metabolite responses of Atlantic cod and Atlantic haddo ck (*Melanogrammus a eglefinus*) sampled from the North Sea close to oil drilling and production platforms. Fish captured close to the platforms also showed enhanced activities of antioxidant enzymes, changes to fatty acid composition, increased concentration s of arachidonic acids and reduced concentrations of Vitamin E (antioxidant), indicating not only accumulation and metabolism of petroleum hydrocarbons but also signs of oxidative stress. Elevated concentrations of DNA adducts in haddock suggested a reacti on of PAH metabolites with double bonds in DNA, mutagenicity and a potential for carcinogenicity. Oxidative stress was also suggested by increased non -enzymatic antioxidant capacity in serum of Gulf killifish (*Fundulus grandis*) exposed to MC-252 oil (Crowe et al. 2014).

Effects of PAHs and oil exposure on the development and function of the embryonic heart should impair the capacity of survivors to swim, capture prey and avoid predation. Effects on embryonic circulation persist into the juvenile stages of fish, a latent effect evident long after exposures to oil. Embryos of mahi-mahi (*Coryphaena hippurus*) that survived 48 hour exposures to weathered MC-252 oil (1.2 µg/L TPAH) showed a reduced critical swimming speed in swim tunnels when tested 24 days after exposure (Mager et al. 2014). Fish exposed to oil (30 µg/L TPAH) for 24 hours as juveniles were 30 -fold less sensitive than fish exposed as embryos. For both groups, no changes occurred in metabolic rate or respiratory efficiency (Mager et al. 2014). Dela yed effects on capacity to swim were also observed in adult zebrafish 10 to 11 months after exposure as embryos to ANS oil (24 -36 µg/L TPAH), a concentration that caused 10 -15% mortality of embryos during exposure (Hicken et al. 2011).

Exposure of fish to crude oil is also associated with impaired immunocompetence and an increased susceptibility to disease, likely due to the effects of PAH s on cellular and humoral immune responses (Reynaud and Deschaux 2006) . However, relating the prevalence of disease to o il exposure in field studies is complicated by the number of uncontrolled environmental stressors that act on fish simultaneously or in different sequences and that can affect immune responses. For example, a variety of hypotheses were examined to explain the crash in abundance of Pacific herring in Prince William Sound following the 1989 EVOS (Pearson et al. 1999), including: a direct effect of oil exposure on survival and

[1] *Not to be confused with the term ;biomarkers' used to describe internal hydrocarbon standards in oil analyses; Chapter 2*

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recruitment of larvae; an indirect effect of oil on the food web supporting herring; the population outgrew its food supply; an epizootic of viral hemorrhagic septicemia (VHS) was triggered by environmental conditions, an over-abundance of herring, impaired immune function, and the stress of oil exposure; overharvesting; and natural chaotic population dynamics. Pearson et al. (1999) concluded that there was no clear link between the decline in herring abundance and oil exposure and that an array of natural factors contributed to the decline. This view was reinforced by Marty et al. (2003) who could not establish a temporal link between epizootics and the 1989 oil spill. In contrast, Thorne and Thomas (2007) related post-spill epizootics in herring with their surfacing behaviour and interaction with oil slicks at the time of the spill, genetic abnormalities that corresponded to oil toxicity and trends in fishery statistics and distribution of herring predators (sea lions). They concluded that the collapse of the herring fishery was a five-year event triggered by the oil spill.

The immune response to pathogens is also complex and can lead to different interpretations of cause and effect, depending on circumstance. For example, when juvenile Pacific herring were exposed to different concentrations of oil in water and their immune systems were challenged by subsequent exposures to a pathogenic bacterium that causes vibriosus, their response depended on duration of oil exposure (Kennedy and Farrell 2008). After one day of exposure, resistance to infection was enhanced, while the opposite occurred after 57 days of exposure. Resistance was associated with a short-term physiological stress response while long-term sensitivity was associated with impaired ion regulation.

Fish diseases can also enhance sensitivity to oil exposure. The cumulative mortality of juvenile flounder (*Pseudopleuronectes americanus*) increased with an eight-week exposure to sediments contaminated with Hibernia crude oil (100 to 2,200 µg/g). At each sediment concentration, mortality was greater in fish infected by the hemoprotozoan, *Trypanosoma murmanensis*, relative to un-parasitized controls (Khan 1991).

During the DWH oil spill, unusual external lesions on red snapper (*Lutjanus campechanus*) in the northern Gulf of Mexico were associated with several pathogenic bacteria. Arias et al. (2013) proposed a causal association between lesion prevalence and exposure to oil. They assumed that the affected fish were immunocompromised by oil exposure, corresponding to previous experience with other fish species (reviewed by Dupuis and Ucan-Morin 2015). However, no evidence was collected of actual exposure to oil, so this proposed cause-effect relationship was not well supported.

<u>Recommendation:</u> Research is needed on the cause of epidemics in fish and the interactions among pathogens, environmental stressors, exposure to oil and prevalence and severity of disease, including chemical and biological markers of oil exposure and effects.

4.1.4.5 Gene mutations and cancer

Mutagenicity and carcinogenicity are often linked to metabolism of specific PAHs. It is well-established that oxygenation of benzo[a]pyrene (BaP; Table 2.1), a pro-carcinogen, by CYP1A enzymes can generate an array of non-carcinogenic and carcinogenic metabolites (e.g., diol-epoxides of BaP) (reviewed by Shugart 1995). Carcinogenic metabolites cause gene mutations by covalent bonding to cellular DNA and interfering with normal replication.

Non-alkylated, pro-carcinogenic PAHs, such as BaP are found at measurable concentrations in crude and refined oils (Yang et al. 2011) but they comprise much less than 15% of total PAHs (Wang et al. 2003). Although methyl phenanthrenes are demonstrated mutagens in cell culture assays (Lavoie et al. 1981, 1982), the mutagenicity of most alkyl PAHs is not well characterized, primarily because few of the thousands of congeners of alkyl PAHs are commercially-available for testing. No papers were found that report carcinogenicity of alkyl PAHs to fish.

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<u>Recommendation:</u> Research is needed to assess the mutagenicity and carcinogenicity of alkyl PAH s to fish.

There has been a variety of studies reporting mutagenicity in fish and invertebrates following marine oil spills (e.g., Bolognesi et al. 2006), but none so far have associated elevated rates of cancer in fish with exposure to oil. Cancers are most evident in benthic species (sessile, in constant contact with bottom sediments, feed on benthic organisms), such as English sole (*Pleuronectes vetulus*; marine) (Myers et al. 2003) and brown bullhead (*Ameiurus nebulosus*; freshwater) (Baumann et al. 1996). Benthic s pecies experience a longer and more intense exposure to potential carcinogens than those in the pelagic zone. The absence of PAH-related cancers in fish following oil spills may reflect the slow onset of chemically -induced cancers (months to years; Baumann et al. 1996), the even slower appearance of frank tumours and the lack of long-term monitoring of fish populations at sites of major oil spills. Interpretation of data on cancer prevalence in fish is also confounded by:

- The age of fish sampled at contaminated and reference sites (prevalence increases with age);
- Co-exposure to other carcinogens not derived from oil;
- Migration of fish to and from sites of exposure; and
- Cell proliferation in response to infections by viruses, bacteria and parasites (Baumann et al. 1996).

In contrast, at sites where high concentrations of pyrogenic PAH s predominate in sediments due to urban and industrial development, there is clear evidence of cancer in locally -resident fish (Myers et al. 2003). Cancer is characterized by pre mature mortality of older fish and corresponding changes in the age structure of the population. However, overall abundance may not change because sexual maturation and reproduction precede mortality (Collier et al. 1992; Bauman et al. 1996). The discovery of tumours in fish exposed to pyrogenic PAH s may be related to the higher concentrations of carcinogenic PAH s, such as BaP and nitrogen heterocycles, and to the long -term intensive studies typical of sites affected by urban and industrial pollution.

<u>Recommendation:</u> Long-term monitoring of impacts following an oil spill should include histological examinations of fish to assess the prevalence of cancer and other diseases in relation to oil exposure.

4.1.4.6 Population and ecosystem consequences of oil toxicity

The effect of oil exposure on cardiac development, structure and function in fish embryos with subsequent impairment of swimming stamina (reviewed in Section 4.1.4.4) indicates that population level effects on fish will follow an oil spill due to a decreased a bility of juveniles to capture prey and avoid predation. The ecological relevance of embryo toxicity is the potential for reduced rates of growth, survival, abundance and productivity. These effects would not be predicted from standard embryo toxicity test s because they are delayed and evident only when the fish are free -swimming, months or years following exposure. For example, Heintz et al. (2000) observed a 15% reduction compared to controls in the rate of return of pink salmon released to the Pacific Oc ean following sublethal exposures to oil as embryos. In contrast, pink salmon exposed to sublethal concentrations of oil as juvenile fish showed no change in returns as adults, compared to controls (Birtwell et al. 1999). The different outcomes of these two similar studies are consistent with the observed high sensitivity to oil of the developing heart of embryos and the much lower sensitivity of juveniles (Mager et al. 2014).

The behaviour and migration of salmon can also be affected by exposure of adults. Although Brannon et al. (1986) found no change in the ability of healthy adult chinook salmon to return to their home stream after a brief (one hour) exposure to Prudhoe Bay crude oil (PBCO), adult coho salmon (*Oncorhyncus kisutch*) avoided a salmon ladder contaminated with oil (Weber et al. 1981). The numbers climbing the

ladder decreased in linear proportion to increasing oil concentrations. *Avoidance is good news from the perspective of survival, but bad news if salmon do not complete their spawning migration and reproduce.*

<u>Recommendation:</u> Further research is needed to determine whether poor returns of adult pink salmon that survived exposure to oil as embryos are related to changes in survival at sea, swimming capacity or response to environmental cues that guide migration.

Despite the apparent high sensitivity of pink salmon embryos to oil exposure in the lab (Heintz et al. 1999), *in situ* (reviewed in Ballachey et al. 2014) and during migration following exposure to oil as embryos (Heintz et al. 2000), t he case for an effect of the EVOS on pink salmon populations in Prince William Sound remains controversial. Population modeling (Geiger et al. 1996; Heintz, 2007) suggested significant impacts of low -level exposures to oil -derived PAH s. However, the technical basis for characterizing impacts and linking oil exposure to effects on recruitment and abundance was questioned by Brannon et al. (2012), mainly because of the complexity of ecological interactions with the population dynamics of sa lmon and the absence of clear evidence of a decline in natural production of salmon (Brannon et al. 2006). Hatcheries and tributaries to Prince William Sound produced so many juvenile fish annually that deficits in production in tributaries affected by the spill were not easily detected. However, the opposite would occur if an oil spill coincided with the spawning of a rare or threatened species, such as the eulachon (*Thaleichthys pacificus*), in the estuaries of west coast rivers (Hodson et al. 2011).

For the DWH oil spill, there were surprisingly few reports of impacts on wild fish at sea or along the coast of the Gulf of Mexico despite numerous laboratory studies demonstrating effects of oil exposure on fish embryos and adults. Recruitment of fish to coast al marshes of Alabama appeared unaffected, as indicated by no change in marsh -associated resident or transient nekton (all actively -swimming aquatic species) over a two -year period following the spill (Moody et al. 2013). However, there were no indicators of potential exposures to oil, other than location. Similarly, the survival of fish embryos in coastal seagrass habitat appeared unchanged in 2010 following the DWH oil spill, in comparison with a previous five-year dataset (Fodrie and Heck 2011). Recruitment of red snapper on artificial reefs was more strongly correlated to dissolved oxygen concentrations and the presence of predators (age -1 fish) than to exposure to oil, and there was no evidence of year -class failure (Szedlmayer and Mudrak 2014). Similarly, a comparison between satellite -based estimates of oil distribution and the modeled spawning habitats of blue-fin tuna indicated that less than 10% of offshore spawning grounds, and 12% of the larval distribution area, would overlap with the distributio n of oil slicks (Muhling et al. 2012), suggesting little impact on recruitment.

Fodrie et al. (2014) reviewed the lack of correspondence between observed effects on fish populations and expectations of impacts from laboratory toxicity studies. They cited several major factors that obscure population-level responses: high spatio -temporal variability in population demographics; movement of fish into and out of affected areas (natural 're -stocking'); fisheries closures that removed a major source of harvest mortality and that compensated for oil-related losses of fish; food web compensation (enhanced productivity due to input of oil -derived carbon); and delayed effects over several generations. Other factors could dampen population responses, such as behavioural avoidance of oil exposure, rapid dilution of oil in the open ocean (particularly following chemical dispersion) and population compensation, such as density-dependent changes in egg, juvenile and adult survival. As well, expectations of effects may be heightened by potential high -sensitivity bias in laboratory tests, depending on experimental designs. Without an obvious fish kill following a spill, as was the case with the Pine River in northern British Columbia (see below) or Asher Cre ek, MO (reviewed by Dupuis and Ucan -Marin 2015), it is very difficult to make an unambiguous case for sublethal, delayed or long -lasting effects. *Nevertheless, numerous laboratory studies (Appendix D, Table D.1) leave little doubt about the potential toxic ity of spilled oil. Difficulties in making the case for oil spill impacts in situ do not mean that population effects do not occur; rather, highly variable population dynamics prevent firm conclusions. Overall, a potential bias in studies of effects on fish populations appears to favour 'false negatives' (no effects perceived on*

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fish populations when there were effects) rather than 'false positives' (effects perceived that were non-existent or due to other factors). These uncertainties need to be resolved and the capacity to define population-level impacts improved.

<u>Recommendation:</u> (adapted in part from Fodrie et al. 2014): Research is needed to: 1) establish an observation network to provide pre-spill baseline data on physical, chemical and biological characteristics of aquatic ecosystems, including fish population, abundance and productivity, particularly in areas where spills are likely; 2) determine individual-level measures of genomic, physiological and demographic responses in baseline surveys as indicators of overall health, rates of disease and exposure to oil and other contaminants (e.g., Stagg et al. 2000); 3) develop a better understanding of the basic ecology and early life-history of fishes including movement, diet, habitat use and longevity and; 4) establish the links between hydrocarbon exposure of individuals and their molecular and cellular responses to signs of toxicity and pathology.

In freshwater ecosystems, EIAs and studies of the rate of recovery of oiled ecosystems are also limited by a lack of data, appropriate assessment methods, and the challenges of sunken oil. There are few field studies of the effects of spilled oil on freshwater fish, other than counts of the dead and dying immediately following a spill. The weakness of short-term census data is that counts of dead fish are often inaccurate because of a low counting efficiency in wilderness areas that are hard to access. Following a spill of sour crude into the Pine River, BC, in 2000, only a partial count of dead fish (1,600) was reported because the river was difficult to access quickly. Dead fish either decayed or were consumed by scavengers before they could be counted. As a consequence, estimates of the total kill varied widely, from 25,000 to 250,000 (Hodson et al. 2011).

Monitoring of time-to-recovery of oiled ecosystems relies too heavily on standing stocks of fish. Without measuring fish population demographics, community structure, movements and productivity (e.g., rates of reproduction and recruitment), actual recovery cannot be separated from 're-stocking' by fish from un-oiled regions of a watershed. Only small amounts of sunken residual oil in sediments could significantly reduce embryo survival and recruitment of sediment-spawning fish species. Following the 2005 spill of HFO 7102 to Wabamun Lake, AB (Section 4.1.2), most residual oil was trapped in shoreline reed beds (Hollebone et al. 2011), but some was measured in offshore spawning shoals of lake whitefish (*Coregonus clupeaformis*) (Short 2008). Debruyn et al. (2007) measured the survival and prevalence of deformities typical of oil exposure of whitefish embryos held in egg containers installed on contaminated and reference shoals in November 2005, four months after the spill. In April 2006, the prevalence of severe skeletal deformities typical of oil exposure was significantly higher among hatched survivors at the exposed site than at a reference site. The rates of pathology were correlated to the amount of TPAHs accumulated by passive samplers installed adjacent to the egg containers.

<u>Recommendation:</u> Research is needed to: 1) develop methods to find and measure residual oil in reed bets and bed sediments of lakes and rivers; 2) assess oil exposure and toxicity of residual oil to embryos of fish species with different spawning media (e.g., sediments, vegitation, water column); and 3) measure the density, survival and recruitment of naturally-spawned fish embryos in oil-contaminated ecosystems.

Some of these research needs may be met by studies of actual 'spills of opportunity'. The current understanding of the ecological effects of oil in marine and coastal environments is derived largely from studies of the 1989 EVOS. Innovative research on the distribution, effects, and persistence of oil in Prince William Sound is cited throughout this chapter and clearly informs current research on the 2010 DWH spill. For Canada, the legacy of the extensive research on these two spills is a much stronger capacity for ERAs and EIAs and the opportunity to advance our knowledge by building on a solid technical base of understanding.

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Much could also be learned if there were small areas of contaminated sites 'set aside' for natural attenuation of spilled oil. A well-known example is a segment of contaminated shoreline untouched 30 years following the *Arrow* spill of Bunker C in Nova Scotia (Lee et al. 2003a). *However, the greatest potential for understanding ecological effects of oil pollution may be studies of oil spilled to experimental marine and freshwater ecosystems. Small-scale experimental spills allow the fewest practical limitations to field investigations and the greatest opportunity for research where gradients of natural environmental stressors can be controlled . To date, the few experimental spills in Canada have been extremely informative (e.g., Sergy and Blackhall 1987; Hodson et al. 2002; Venosa et al. 2002; Garcia Blanco et al. 2007), but their scopes were limited by the questions being asked. Additional experimental spills are needed to keep pace with Canada's rapidly expanding oil trans portation network and the risks to a wide array of ecosystem types.*

4.1.5 Reptiles and amphibians

No species of sea turtles nest in Canada, but five of the world's seven species are sighted on an occasional basis on both the east and west coast, including green (*Chelonia mydas*), Atlantic loggerhead (*Caretta caretta*) (and a hybrid of the two), Atlantic ridley (*Lepidochelys kempii*), olive ridley (*L. olivacea*) and leatherback (*Dermochelys coriacea*) turtles (www.thecanadianencyclopedia.ca/en/article/turtle/). *While several studies of turtles in the northern Gulf of Mexico were published before and after the DWH oil spill, no links were established among the status of different species, indicators of health and exposure to oil* (Beyer et al. 2015). However, all studies identified a high risk of severe damage to populations should there be contact with oil and emphasized the need for conservati on efforts to sustain populations that are under pressure from a variety of anthropogenic stressors.

Native species of turtles, other reptiles (43 species total) and amphibians (46 species) inhabit terrestrial and freshwater habitats of southern Canada, wi th their northern distribution limited by the boreal forest . No species inhabit the tundra zone (www.thecanadianencyclopedia.com/en/). Oil spills to aquatic ecosystems and riparian lands will crea te significant risks of exposure of reptiles and amphibians to petroleum hydrocarbons, as well as potential impacts on their habitats from oil spill cleanup. Turtles may be coated with oil or inhale volatiles or aerosols of oil when they surface and the eg gs of species that spawn on shore may be exposed to stranded oil. Numerous studies have demonstrated that turtles, particularly long -lived predatory species at a high trophic level, bioaccumulate persistent organic pollutants and develop dose-dependent deformities (Rowe et al. 2009). However, the evidence for effects of petroleum hydrocarbons, particularly PAH s, on turtles is less established. Bell et al. (2006) reported deformities in snapping turtles (*Chelydra serpentina*) and painted turtles (*Chrysemys pi cta*) from a wildlife refuge in Pennsylvania that was heavily contaminated by past industrial activity with metals, PAHs and other organic compounds. The prevalence and severity of deformities far exceeded those at a reference site, and were equivalent to or greater than those reported at other contaminated sites. Although measured concentrations of PAHs in turtles at the contaminated site were far greater than at the reference site, the cause of deformities was not conclusive because the study had insufficient resources to analyze metals and other organic compounds . At this same refuge, the survival, home -range and water temperature preference of four species of freshwater turtles contaminated by an oil spill were unaffected relative to un-oiled turtles, despite their capture, rehabilitation by cleaning and release (S aba and Spotila 2003). There was a similar experience following the 2010 spill of dilbit to Talmadge Creek and the Kalamazoo River. Large numbers of juvenile and adult turtles were rescued, cleaned and released (USFWS 2015) and because some were captured and released up to five times, it was concluded that they were unharmed by the experience.

Although the short -term effects of oiling on free -swimming turtles appear negligible, there were no studies of the chronic effects of the dilbit spill or of the effects on the embryonic stages of turtles exposed to dilbit on the exterior of egg shells. For Arabian Light crude oil, eggs of snapping turtles showed a very low sensitivity to un -dispersed oil (WAF) or to chemically -dispersed oil (chemically-enhanced WAF or

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CEWAF) (Rowe et al. 2009). After single doses of WAF or CEWAF were percolated through nests containing viable eggs, there were no significant effects on hatching success, weight, length, lipid content, DNA damage or overall survival post-hatch. The absence of effects corresponded to tissue concentrations that were less than those observed in deformed turtles from a contaminated site (Bell et al. 2006) . In contrast, eggs of loggerhead turtles buried in nests on a Florida beach were killed by a spill of fuel oil, although the cause may have been suffocation, not toxicity (reviewed in Rowe et al. 2009).

The unique adaptations and life histories of some species, such as beach nesting by marine turtles, may create unique sensitivities to oil exposure and effects. A potential cause of mass die-off of marine iguanas (*Amblyrhynchus cristatus*) following a diesel spill in the Galapagos Islands may have been due to effects on algae, their primary diet, or on their unique gut symbionts (Wikelski et al. 2011). In fresh water, amphibians (frogs, salamanders) that spawn in clear, shallow ponds may be subject to photo-enhanced toxicity (Section 4.2.5). The toxicity of a diluent recovered from contaminated groundwater in California to tadpoles of the southern leopard frog (*Rana sphencephala*), increased by three-fold in the presence of UV-B at wavelengths and intensities typical of California (Little et al. 1998).

These limited studies reviewed are insufficient to draw firm conclusions about the relative risks of toxicity to turtles and amphibians exposed to oil. The influence of variables , such as exposure regimes (single doses vs. repeated doses or continuous exposures), species characteristics (e.g., the nature of the egg shell and chorion and permeability to hydrocarbon uptake) and the relative toxicity of LMW and HMW PAHs, remain unknown. Of particular importance in Canada are the interactions between oil exposure and the natural stressors acting on turtles that are at the northern limit of their range. Extremes of temperature and short seasons for growth and reproduction could aggravate even minor effects of oil toxicity.

Recommendation: Research is needed on the toxicity of oil to embryonic and juvenile turtles and amphibians to determine which species characteristics contribute most to sensitivity to oil and which exposure routes are most harmful.

4.1.6 Birds

Aquatic birds are at risk of acute mortality following a spill if their feathers become contaminated . Bird feathers have a function in flight, water repellency and thermoregulation, and all three functions are significantly impaired by contact with oil. In severe cases, birds are unable to fly , at risk of drowning because of loss of buoyancy and at risk of hypothermia and starvation due to heat loss, accelerated metabolism to generate heat and a reduced ability to feed. Issues of thermoregulation are particularly acute if birds are:

- Oiled during winter months (marine coastal zones, Great Lakes);
- In areas influenced by currents of cold Arctic water (e.g., Gulf of St. Lawrence);
- At higher latitudes (seasonal use of Arctic islands for breeding and nesting); or
- On migration flyways in spring and fall.

Oil spills in areas frequented by aquatic birds often kill hundreds of thousands of birds. For example, the mean mortality (with 95% confidence limits) of over-wintering birds in the Gulf of Mexico during the DWH oil spill was estimated as 600,000 birds (320,000 to 1,200,000) using a carcass sampling model, and 800,000 birds (160,000 to 1,900,000 using an exposure probability model (Haney et al. 2014a, b). The most affected species were predicted to be laughing gulls (*Leucophaeus atricilla* , 32% of the population), royal terns (*Thalasseus maximus* , 15%), brown pelicans (*Pelecanus occidentalis* , 12%) and northern gannet (*Morus bassanus*, 8%). A potential source of bias in the sampling model was the low probability of recovering carcasses of oiled birds during coastal surveys. In a stranded bird survey, most of the 7,229 dead birds collected were found near the coast adjacent to New Orleans and the stricken oil

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platform (Tran et al. 2014), reflecting the source and distribution of oil during the spill. Shorebirds may be similarly affected and in the Gulf of Mexico it was estimated that 28 species, comprising more than 1,000,000 birds, were at risk of exposure during the DWH spill (Henkel et al. 2012). For the dunlin (*Calidris alpina*), a shorebird, 8.6% and 0.6% of live birds showed light to trace oiling in 2010/11 and 2011/12, respectively, indicating that as many as 100,000 birds could have been affected by oil (Henkel et al. 2014). This number would be even higher without cleanup operations (Beyer et al. 2015).

Similar effects were evident in northern marine ecosystems affected by oil spills, particularly if the spill coincided with breeding. In Prince William Sound, the estimated number of direct mortalities of seabirds (with 95% confidence limits) following the EVOS was about 375,000 (300,000 to 645,000), based on the numbers of oiled carcasses recovered and an intensive study of the proportion of carcasses that disappeared due to scavengers before being enumerated (Piatt and Anderson 1996). The estimated half-life of beached carcasses was ≈1 day and systematic bird surveys only began two weeks after the spill (Ford et al. 1996). Early in the spill, 100% of recovered carcasses were oiled, but the percentage oiled declined to zero within four months for a wide array of species. While acute mortality of birds, such as the common murre (*Uria aalge*; comprised 70% of beached birds), could be attributed to oil exposure, longer-term effects on abundance were much less certain due to ecosystem-wide changes in abundance and species composition of prey (fish, invertebrates) in Prince William Sound (Piatt and Anderson 1996).

Local effects of oil spills may also have continent-scale impacts. Species such as the northern gannet, common loons (*Gavia immer*) and a variety of shorebirds migrate to and from summer nesting sites in the Arctic, inland lakes and both coasts of Canada. Mortality during incidents such as the DWH oil spill may have continent-wide effects on population dynamics and abundance. Monitoring is needed to establish baseline estimates and trends in abundance and productivity of nesting populations and to assess whether oil spills in over-wintering areas or along major flyways have affected one or more species. For example, after the DWH spill, new software tools, such as Google Fusion Tables and Google Maps, were applied to an existing wildlife database to show whether the distribution of aquatic birds overlapped the area affected by the spill (Tran et al. 2014).

Sublethal oiling of birds can also affect reproduction. Even a few microlitres of oil transferred from the parent to the surface of eggs during incubation can be absorbed and cause deformities and embryo mortality. The median lethal dose (LD50) of weathered MC-252 oil applied to the exterior of mallard eggs (*Anas platyrhynchos*) was 0.5 µL or about 0.5 mg oil/egg (Finch et al. 2011). In contrast, dispersed oil was four to six times less toxic than un-dispersed oil, suggesting that the dispersant mitigated oil toxicity, perhaps by enabling a more rapid loss of oil from the egg's surface. However, Corexit® 9500 by itself at doses of 100 µL/egg caused acute mortality and lower doses reduced hatching success (Finch et al. 2011).

In areas of Prince William Sound affected by the EVOS, reproductive rates appeared lower for birds that feed in open water (e.g., kittiwake [*Rissa tridactyla*]) (Irons 1996) and on shorelines (e.g., black oystercatcher [*Haematopus bachmani*]) (Sharp et al. 1996). Relative to un-oiled areas, winter survival rates of female harlequin ducks (*Histrionicus histrionicus*) were reduced for up to 10 years following the spill (reviewed in Ballachey et al. 2014). Effects on pigeon guillemots (*Cepphus columba*) were less obvious due to the difficulties in finding well-scattered and concealed nests, but a general decline in abundance seemed to be exacerbated by the spill (Oakley and Kuletz 1996).

Understanding the impacts of oil spills on aquatic birds is limited by the small number of comprehensive surveys during oil spills and the difficulties in recovering oiled birds before they disappear. For example, in busy shipping routes off Newfoundland, the disposal of waste oil and contaminated bilge water causes a chronic and continuous loss of seabirds, such as thick-billed murres (*Uria lomvia*), as indicated by winter surveys of 13 beaches on the Avalon Peninsula (Wiese and Ryan 2003). Although deaths related to oiling have been increasing, the total number of birds killed could not be estimated from these data. Only

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a small fraction of the Newfoundland shoreline adjacent to shipping routes could be surveyed, and the proportion of affected birds that did not strand on shore was unknown. However, trends were validated by significant negative correlations between the percentage of murres oiled and the annual harvest, implying an impact of mortality on overall abundance.

For freshwater spills, data are not readily available for effects on aquatic birds. To cover damage assessment needs, Irons (1996) recommended the immediate collection of data on oil exposure, distribution, abundance and behaviour of birds at spill sites where baseline data is lacking. Dead fish and birds were observed in shoreline reed beds following the 2005 spill of heavy fuel oil to Wabamun Lake, AB (Birtwell 2008; Section 4.1.2), but estimates of total numbers have not been published. *The ecological significance of widespread bird mortality from oiling is also poorly understood.* Experience with the DWH spill indicates that widespread mortality of fish-eating birds may have had 'top-down' effects on the marine ecosystem of the Gulf of Mexico, changing the abundance and age structure of fish populations that supported the birds (Short 2015).

<u>Recommendation:</u> Research is needed to: 1) validate methods for estimating bird numbers and their susceptibility to oil spills; and 2) establish long-term pre-spill baselines of population abundance and an understanding of food web functioning and productivity in areas of Canada at risk of oil spills, particularly where birds over-winter (e.g., the Great Lakes), along major migratory flyways or in nesting colonies.

4.1.7 Mammals

Aquatic mammals are particularly susceptible to floating oil because they must surface at regular intervals to breathe. Some species can sense the presence of oil, and captive bottlenose dolphins (*Tursiops truncates*) have even been trained to detect and avoid oil (Engelhardt 1983), but field observations indicate that most species do not consistently avoid spilled oil; within 24 hours of the EVOS, killer whales were swimming through surface slicks (reviewed by Dupuis and Ucan-Marin 2015). Even if mammals could detect oil, exposure may be unavoidable in Arctic ecosystems if ice can herd floating oil into thick layers in breathing holes and leads between ice floes (Chapter 2). When mammals surface through oil, their skin or fur will be contaminated, and they may inhale liquid oil or oil aerosols during rapid, high-volume respiration, causing lung damage or narcosis and drowning. For filter-feeding baleen whales, exposure includes consumption of oil-contaminated plankton and, in extreme cases, baleen may be clogged by liquid oil (Engelhardt 1983).

Aquatic mammals that rely on fur for insulation face the same issues of acute mortality due to thermoregulatory failure as do birds. Only a light oiling may lead to serious heat loss and the risk of death by hypothermia or starvation. All species are not affected equally. For species that do not have fur (e.g., cetaceans), the significance of direct contamination of skin is less clear and, in fact, oil may not stick as readily to such species (Engelhardt 1983). Seals accumulate oil on fur, but hypothermia may be offset somewhat by thick layers of blubber, although there is no information about newborns. Sea otters (*Enhydra lutris*) do not have blubber for thermal insulation. If oiled, they must increase their time spent foraging to compensate for an increased rate of metabolism due to stress and thermal regulation. However, otters groom compulsively to maintain the water repellency and thermal insulation of their fur and may starve if time spent grooming increases at the expense of time spent foraging (reviewed by Sinclair 2012). Some otter populations in Prince William Sound experienced elevated mortality rates for up to 22 years following the EVOS. Mortality was associated with intermittent exposure to residual oil in intertidal sediments where otters forage for clams (reviewed in Ballachey et al. 2014).

Hydrocarbons consumed by aquatic mammals through grooming or contaminated diets can be metabolized and readily excreted, but some is stored in blubber and other fat deposits. Engelhardt (1983) speculated that stored hydrocarbons may be released into circulation when fat is consumed during periods

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of physiological stress (e.g., low food availability, migration, lactation), and circulating hydrocarbons may be bioavailable and toxic to the fetus or nursing newborns. Tissue pathology identified by necropsies of animals recovered after oil spills included:

- Irritation of tissues around the eyes;
- Kidney pathology that may reflect the excretion of hydrocarbon metabolites;
- Hemolysis, anemia and erythropoietic dysfunction;
- Liver and kidney dysfunction, including uremia and dehydration;
- Lesions in the upper digestive and gastro-intestinal tracts; and
- Changes to the adrenal cortex suggesting significant responses to stress and increased kidney function (Engelhardt 1983).

Mortality of a variety of sea mammals has been frequently associated with marine oil spills. Sea otters, harbour seals (Phoca vitulina), Stellar sea lions (Eumatopius jubatus), killer whales (Orcinus orca) and humpback whales (Megaptera novaeangliae) were most affected by the Exxon Valdez oil spill. Elevated rates of mortality were evident in the months following the spill in oiled areas compared to un -oiled (Loughlin et al. 1996). Higher concentrations of hydrocarbons in carcasses of ott ers, harbour seals and sea lions from oiled areas strengthened the connection between oil exposure and effects.

As with fish, links between oil exposure and acute and chronic effects remain controversial because most marine mammals are quite mobile and not easily studied. Mortality may not occur in the immediate vicinity of an oil spill, particularly if oil causes slow starvation due to chronic respiratory damage, disruption of thermal regulation or an inability to feed properly. For example, the estimated carcass recovery rates of cetaceans after the DWH spill were as low as 2% (Williams et al. 2011), which limit s the statistical validity of any proposed cause-effect relationships. Without direct evidence of an exposure to oil, it may be impossible to discr iminate oil toxicity from multiple other causes of morbidity. During the DWH oil spill, there was a prolonged 'Unusual Mortality Event' (UME) of bottlenose dolphin in the Gulf of Mexico (Carmichael et al. 2012a) that began before the oil spill and persiste d for months afterwards. The UME overlapped in time and space with the spill, and dolphins were observed swimming and feeding in areas of oil-contaminated waters (Schwacke et al. 2014). Although the UME began before the spill, its prolongation and severity were attributed to oil exposure, implying interacting and delayed toxicity. Thinning of the adrenal cortex (hypoadrenocorticism, suggestive of chronic stress) combined with bacterial pneumonia, the primary cause of death, corresponded to previous observat ions of poor health in mammals exposed to other oils. Dolphins from oil-exposed areas were five times more likely to have moderate-to-severe lung disease than dolphins from an un -oiled reference site. Venn-Watson et al. (2015) strengthened the case for oil toxicity by examining dead and stranded dolphins collected from oiled and reference sites. They confirmed the high prevalence of bacterial pneumonia and hypoadrenocorticism, but eliminated algal toxins, morbillivirus infection and brucellosis as contribut ing factors. Similarly, Litz et al. (2014) eliminated morbillivirus and brevitoxicosis as causes of the UME. Although the proposed cause -effect relationship was biologically plausible and the effects corresponded in timing and location to possible oil exposure, the case for oil toxicity was weakened by a lack of data on concentrations of hydrocarbons or hydrocarbon metabolites in tissues, urine or bile. Similarly, there were no measurements of molecular or biochemical markers of hydrocarbon exposure or toxicity , such as elevated concentrations of CYP1A proteins or DNA adducts (Fossi and Marsili 1997). An analysis of external wipes from 13 of 35 beached bottlenose dolphin carcasses collected between May 2010 and November 2012 found residues typical of weathered Macondo oil (Stout 2015), indicating exposure at the time of the spill and ongoing exposure over the next two years. Low concentrations of hydrocarbons in digestive-tract materials were not consistent with dietary exposure. These data s upport the role of oil exposure in the UME event, but the cause-effect case would be stronger if data on pathology and contamination were linked for each stranded animal.

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During the DWH spill, skin biopsies of sperm whales (*Physeter macrocephalus*) sampled from the Gulf of Mexico demonstrated nickel and chromium concentrations two to five times higher than background (Wise et al. 2014). Concern was expressed for potential genotoxicity , but the form s of nickel and chromium in MC -252 oil residues were not measured (the valence state determines carcinogenicity). Overall, the link s between tissue metal concentrations and exposure to the DHW spill were not clear, given that the whales travelled widely throughout the Gulf , there are other sources of metals in the Gulf, including the Mississippi River , and other measures of oil exposure (tissue concentrations of petroleum hydrocarbons, biomarkers of oil exposure and effect) were not included.

Recommendation: There is a need for more in-depth research on the accumulation, disposition and effects of petroleum hydrocarbons and metals in marine mammals, and methods to link biomarkers of oil exposure and toxicity to specific sources of oil.

The DWH spill may have also affected marine mammal behaviour, as indicated by acoustic recordings of sperm whales that suggested they had moved away from the spill (Ackleh et al. 2012). Information about marine mammal behaviour may also be key to mitigating potential impacts on many species following spills in Canadian waters. For example, the seasonal abundance, migrations and behaviour of humpback whales in the Bay of Fundy, beluga, fin and blue whales in the Gulf of St. Lawrence, and killer and humpback whales off the British Columbia coast are well established. This baseline information has already influenced decisions on major shipping routes in the Bay of Fundy, in this case to avoid collisions, and on the siting of a major oil terminal in the St. Lawrence River estuary to minimize the potential exposure of beluga whales to oil spills.

Recommendation: Monitoring is needed in areas of existing and proposed oil -related activities to ensure that the seasonal and spatial use of habitat by marine mammals is well described and that risks of oil exposure are minimized.

4.1.8 Ecosystem services

The impacts of oil spills in aquatic ecosystems have been viewed traditionally in terms of oil effects on individual species, populations or ecosystems. Ecosystem services provide a different perspective on oil spill effects, focusing on values to human society that are affected, either negatively or positively, by ecological impacts.

Ecosystem services are "the benefits provided by ecosystems to humans that contribute to making human life both possible and worth living" (NRC 2013) and are derived f rom functioning ecosystems, i.e., the interactions among plants, animals and microbes as primary and secondary producers, among predators, prey and competitors, and between living species and environmental characteristics such as temperature and habitat. The physical and chemical properties of an environment also provide services, such as modifying climate by storing heat and carbon dioxide. Services can be classified into four categories (NRC 2013):

- *Provisioning services* (e.g., material goods such as food, feed, fuel and fiber);
- *Regulating services* (e.g., climate regulation, flood control and water purification);
- *Cultural services* (e.g., recreational, spiritual and aesthetic services); and
- *Supporting services* (e.g., nutrient cycling, primary production and soil formation).

Ideally, each service can be valued economically, allowing the damage to aquatic ecosystems from an oil spill to be expressed monetarily and in ways that are easily understood by non -ecologists. For example, when access to an ocean beach i s blocked by an oil spill for several months during the tourist season, the loss of income from tourism can be accounted for in the overall cost of cleanup. An analysis of ecosystem

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services can provide a basis for compensating the public for lost opportunities to use the beach and the businesses or individuals who have suffered financial loss following a reduction in the tourist trade.

This concept may generate a much broader appreciation of ecological, social and economic impacts (i.e., on human well-being), the nature of damages included in a post-spill assessment, and the potential options for remediation and restoration. Its implementation will require a broader and more systematic framework for structuring programs of research and monitoring, for developing comprehensive models of how ecosystem services are generated, delivered and affected, and for identifying the trade-offs and balances in protecting some ecosystem services, potentially at the expense of others (Net Environmental Benefits Analysis). More importantly, analyses of ecosystem services will define the data that must be collected to support the approach, the extent and nature of remediation needed to restore services to their pre-spill values and include the cultural and economic values of affected communities, particularly indigenous communities, to create perspective on the question of "How clean is clean enough?"

A US National Research Council (NRC) Technical Panel provided a comprehensive review of the 2010 DWH oil spill to determine if the ecosystem services concept provided an additional and useful approach for assessing resource damage of the oil and of the spill cleanup under the US Natural Resources Damage Assessment process (NRC 2013). After examining four case studies in detail (wetlands, fisheries, marine mammals and the deep Gulf of Mexico), they identified *three primary obstacles to the comprehensive application of the concept:*

1. *A lack of baseline measurement of goods and services produced by the Gulf of Mexico ecosystems just prior to the harmful event. Only for fisheries of commercially-important species are there adequate data for an ecosystem services analysis because the mandate of fisheries agencies is to track abundance, harvest, price and overall economic activity;*
2. *The technical challenge of developing a comprehensive model to predict the ecological impacts of an oil spill and associated economic impacts. Such models can be developed from existing ecological and economic valuation models, but a major obstacle can be a lack of socioeconomic understanding and data that describe complex human dependencies on natural systems; and*
3. *The challenges of realistic and accepted economic valuations of ecosystem services, particularly for services that are not valuated in market activity (e.g., the value of deep sea sediments as a repository for persistent contaminants from oil spills). If ecosystem services cannot be valued easily, there is a risk that these services could be discounted or ignored in the decision-making process (NRC 2013).*

The relevance and value of the NRC (2013) report was derived from the legal requirements for Natural Resources Damage Assessment under the *1990 US Oil Pollution Act* (NRC 2013).

<u>Recommendation:</u> To benefit from an ecosystem services approach to assessing impacts, research is needed to determine if existing regulatory frameworks for spill response, remediation and monitoring provide an adequate basis for defining and assigning responsibility for its implementation. Comprehensive ecological and socioeconomic models must be developed in collaboration with communities at risk, particularly indigenous communities to evaluate the significance of ecosystem services for marine and freshwater ecosystems.

4.2 Factors Affecting Oil Toxicity to Fish

4.2.1 Species tested

There are considerable differences among species in their sensitivity to oil, some of which relate to their physiology and life history, some to the environmental characteristics of their habitats (e.g., temperature, salinity) and some to differences among laboratories in test methods, experimental designs and responses

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measured. Compilations of toxicity data across a wide range of species can illustrate the relative sensitivity of an individual species (**Appendix D, Table D.1**), identify the more sensitive species and define a benchmark for protecting the majority of species.

Barron et al. (2013) provided a useful perspective on the relative acute lethality of an array of crude and fuels oils to aquatic species, as well as two dispersants tested without oil. Species sensitivity distributions (SSDs) for 227 measures of acute lethality (LC50s) for 67 fish and invertebrate species typical of warm and cold waters were expressed as measured concentrations of TPHs. Data were selected for a mix of marine and freshwater species based on several criteria, including:

> **Species sensitivity distributions (SSDs)** - compare the cumulative proportion of species (percentile) affected by oil to a toxicity endpoint measured for each species (e.g., the 96 hour LC50). SSDs display the range of concentrations associated with a specific effect and can be used to estimate the concentrations that affect a given percentage of tested species. The 'HC5', or hazardous concentration harming the 5th percentile of species, is often used in ecological risk assessments to establish toxicity benchmarks that should, in theory, protect 95% of all species from the effect represented by the endpoint.

- All life stages except embryos;
- Test duration (invertebrates: 48–96 hours; fish: 96 hours);
- Exposure regime—i.e., gradients of oil concentrations administered as a continuous flow of (rare) or as static tests with daily solution renewal (most common);
- Fresh oil only;
- No chemical dispersants; and
- Oil exposure characterized by measured concentrations of TPH.

HC5 values were derived from a curvilinear regression of percentile responses of 10 to 39 species against log LC50 values (mg/L TPH). *Invertebrate species were distributed evenly throughout the SSDs indicating they were not consistently more or less sensitive than fish species, as expected for a non-specific narcosis MOA typical of acute lethality.* One acknowledged bias in this study was the separation of species tested by oil type. As might be expected, northern oils (e.g., ANS crude) were tested more frequently with coldwater species, while southern oils (South Louisiana crude) were tested more frequently with warmwater species. However, for inland silversides, a warmwater estuarine fish, relative sensitivity in each SSD was not fixed but varied with the oil tested and the mix of species included (Barron et al. 2013), suggesting no consistent bias (see also section 4.2.6).

For the chronic toxicity of oil to fish embryos, SSDs were prepared from the data summarized in **Appendix D, Table D.1**, selecting only those studies reporting chronic lethal (LC50) and sublethal (EC50) toxicity values expressed as measured concentrations of hydrocarbons in test solutions (**Figure 4.3**). Some SSDs are incomplete because there were too few data for chronic LC50s expressed as TPAH concentrations, and too few data reporting chronic EC50s expressed as TPH concentrations. *Nevertheless, HC5 values calculated from the regressions of Figure 4.3, or estimated by eye, ranged over two orders of magnitude, from 0.23 to 3.2 µg/L TPAH or 10 to 190 µg/L TPH (**Table 4.1**), and the SSDs indicated no consistent differences between freshwater or marine species in sensitivity.*

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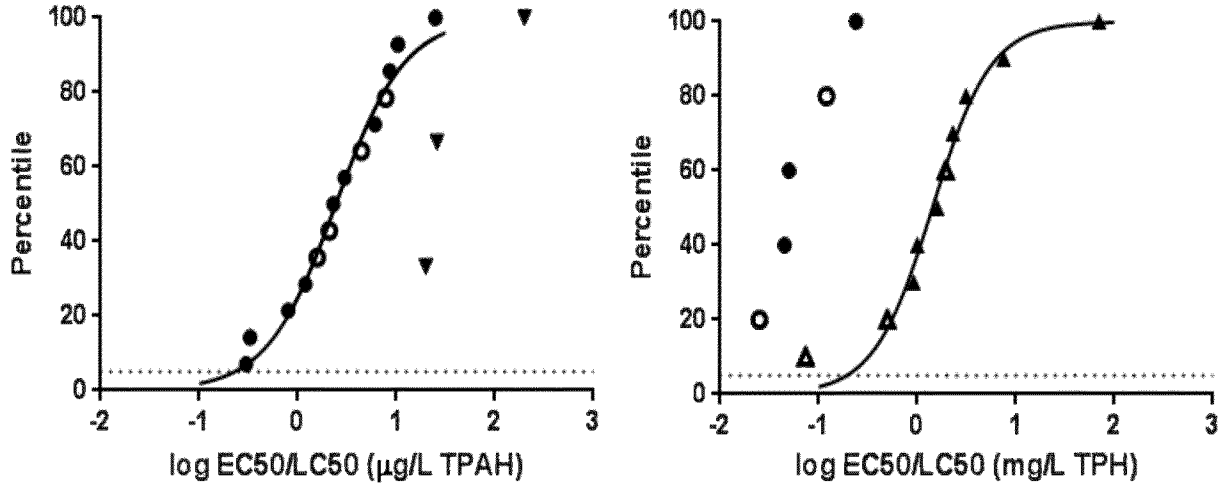

Figure 4.3 *Species sensitivity distributions for the sublethal (circles, EC50 values) and lethal (triangles, LC50 values) toxicities of oil, expressed as total polycyclic aromatic hydrocarbons (TPAH; Panel A) and total petroleum hydrocarbons (TPH; Panel B) to marine (filled symbols) and freshwater species (open symbols). Data were selected as the lowest EC50 or LC50 values for a given species, so that one species did not do minate the analysis. Only datasets with 10 or more species were analyzed statistically with a non -linear regression (GraphPad Prism Ver. 5; GraphPad Software Inc, LaJolla, CA). The horizontal line represents the 5 th percentile of species and intersects the regressions at an EC50 or LC50 equivalent to the 'HC5'. Data were drawn from papers summarized in* ***Appendix D, Table D.1*** *(Kocan et al. 1996; Marty et al. 1997; Carls et al. 1999, 2008; Gulec and Holdway 2000; Little et al. 2000; Brand et al. 2001; Pollino and Holdway 2002; Couillard et al. 2005; Perkins et al. 2005; Shen et al. 2010; Olsvik et al. 2011; Greer et al. 2012;; Wu et al. 2012; Gardiner et al. 2013; Incardona et al. 2014; Mager et al. 2014; Martin et al. 2014; Dussauze et al. 2015; Madison et al. 2015; Sørhus et al. 2015).*

The primary limitations of establishing toxicity benchmarks from SSDs are the assumptions that the species included represent the range in sensitivity of all species in all ecosystems under all environmental conditions, and that the SSD truly reflects differences in toxicity among species and not the selection of test species according to their tolerance of lab conditions or differences in test methods among laboratories. *Nevertheless, SSDs provide a reasonable first est imate of toxicity benchmarks, although a comprehensive comparison among species and oils within one testing laboratory would increase the confidence in their application to risk assessments.*

<u>Recommendation:</u> SSDs should be improved by testing a wider array of test species, with toxicity expressed as TPAH and TPH concentrations, or as concentrations of specific hydrocarbons identified as the toxic components of oil.

Table 4.1 *The concentrations of TPAH s and TPHs estimated to be hazardous to 5% of test species, as estimated from species sensitivity distributions of chronic embryo toxicity (**Figure 4.3**)*

	HC5 calculated from regression equations	HC5 estimated from data distributions
TPAH – EC50 (µg/L)	0.23	
TPAH – LC50 (µg/L)		3.2
TPH – EC50 (mg/L)		0.01
TPH – LC50 (mg/L)	0.19	

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4.2.2 *Type of oil and oil characteristics*

As indicated in Section 4.1, *the acute toxicity of crude and refined oils has been attributed to LMW hydrocarbons that cause lethality by narcosis, regardless of life stage, while MMW 3 -5-ringed PAH are thought to be the primary components that cause chronic toxicity to fish embryos.* Based on this knowledge, it might be expected that the relative risks of acute and chronic toxicity of different oils could be assessed by simply ranking them in order of the concentrations of their toxic constituents. *For example, very light oils that are rich in monoaromatics and short -chain alkanes should be more acutely lethal than heavy fuel oils with smaller proporti ons of these hydrocarbons.* While this may generally be true, the SSDs developed by Barron et al (2013) for acute lethality of crude and fuel oils demonstrated that estimated HC5 values varied widely among oils, from 0.29 to 3.5 mg/L TPH . Light and heavy fu el oils were more acutely toxic (0.29 and 0.56 mg/L TPH, respectively) than four crude oils (0.9 to 3.5 mg/L TPH). The lightest oil (South Louisiana crude) was two - to three-times less toxic than the average for all oils, based on measured TPH concentratio ns. No toxicity data for very light oils, such as Bakken crude, were found.

By the same reasoning, *oils that are enriched in alkyl PAH s (e.g., heavy fuel oils) should cause greater chronic toxicity to fish embryos than oils with low concentrations of alkyl PAHs. Virtually all crude and refined oils tested to date cause a similar array of toxic responses in fish embryos* (**Appendix D, Table D.1**), including light and heavy crudes (Wu et al. 2012), refined products, such as diesel (Schein et al. 2009) and HFO (Incardona et al. 2012a; Martin et al. 2014), and bitumen (Colavecchia et al. 2004, 2006) and products derived from bitumen, such as Orimulsion (Boudreau et al. 2009), and diluted bitumen (Madison et al. 2015). *However, toxicity is highly variable and may b e confounded by the relative proportions of different alkyl PAH in each oil*. For example, Martin et al. (2014) measured the toxicity of a crude and heavy fuel oil to trout embryos expressed as measured concentrations of TPAH s in test solutions (**Table 4.2**). If toxicity was determined solely by TPAH s, there should be little difference between the oils in TPAH concentrations causing embryo toxicity. However, HFO was 4 to 5-times more toxic than crude oil, perhaps due to higher proportions of the more embryotox ic 3 - to 5 -ringed alkyl PAHs, differences in bioavailability of the toxic constituents, or the presence of toxic compounds not typically analyzed in oil , such as polar compounds (e.g., phenolic derivatives of alkyl PAH s, alkyl dibenzothiophenes; reviewed in Section 4.1.4.3).

It is often difficult to express relationships between PAH concentrations in oil and toxicity quantitatively because the physical characteristics of oil (e.g., density, viscosity) modify toxicity by controlling droplet formation and the rate of partitioning of hydrocarbons from oil to water. The proportion of HMW waxes, resins and asphaltenes will control the physical characteristics of oil (Chapter 2), which in turn affect the dispersability of oil in water. Thus, relative to light oil s, heavy oils are more difficult to disperse mechanically in water, are less likely to form fine droplets and may appear less toxic than light oils because their toxic constituents partition more slowly to water.

Weathering of oil changes the relative concentrations of different toxic components of oil, so that toxicity may change over time depending on what compounds are lost by volatilization and oil-water partitioning (e.g., Carls et al. 1999) . Microbial transformations of hydrocarbons (review, Neff et a l. 2013) may also create hydroxylated derivatives that are more toxic than the parent compound (Hodson et al. 2007b; Fallahtafti et al. 2012). Thus, the measured toxicity of fresh oil may over - or underestimate impacts of spilled oil as it weathers (Neff e t al 2000) . This bias may also occur in lab studies due to the risk of thermal degradation of hydrocarbons if oil is heated to simulate weathering.

<u>Recommendation:</u> Microbial metabolites in weathered oil and their contribution to toxicity need to be determined.

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Table 4.2 *A comparison of the concentrations of TPAHs in two test oils to the concentrations in test solutions causing chronic toxicity (25 day EC50 values) to trout embryos (Martin et al. 2014).*

	MESA crude oil	HFO 6303
TPAHs in undiluted oil (µg/g)	13,415	42,586
Median effective concentration (25 day EC50) for sublethal embryo toxicity to rainbow trout (µg TPAH/L)	23	5

Recent spills of diluted bitumen (dilbit; e.g., Kalamazoo River, Chapter 8) have raised concerns about unique threats to aquatic environments because the composition and proportions of hydrocarbons in dilbit differ considerably from conventional crude oils. As outlined in Chapter 2, dilbit is prepared from bitumen, a highly-weathered oil depleted in LMW compounds. When bitumen is extracted from the oil sands, many of the polar and water soluble components are removed by water washing, including naphthenic acids (NAs), naturally-occurring, linear and cyclic carboxylic acids found in the acidic fraction of crude oil and raw bitumen. The NAs account for much of the acute toxicity of oil sands process water. Because extracted bitumen is too viscous to flow through pipeline, diluents such as natural gas condensates are added to reduce viscosity, in effect replacing many of the LMW compounds lost during bitumen formation. The amount of diluent added changes to accommodate seasonal temperature effects on viscosity and the chemical composition changes with its commercial source. As a consequence, there may be differences in the chemistry and toxicity of diluent between summer and winter blends, and among batches.

No publications were found describing the acute lethality of fresh or weathered dilbit to fish. Bitumen itself is chronically toxic to fish exposed to bituminous sediments, as demonstrated by blue sac disease in embryos of white sucker (*Catostomus commersoni*) and fathead minnow (*Pimephales promelas*) and endocrine disruption of slimy sculpin (*Cottus cognatus*) (reviewed in Dew et al. 2015). Alkylnaphthalenes, the PAH of lowest chronic toxicity, are mostly lost during the formation of bitumen, so that Access Western Blend (AWB) dilbit contains a lower proportion than conventional crude oils (Yang et al. 2003). Nevertheless, the *concentrations of 3-5-ringed alkyl PAHs in AWB are equivalent to those of conventional oils, suggesting that chronic toxicity should not differ markedly.* There is only one published report on the chronic embryo toxicity of AWB dilbit (Madison et al. 2015). The EC50 for Japanese medaka fell in the 37th percentile of an SSD for an array of crude and refined oils (**Figure 4.3**). This rank suggests that dilbit may be slightly more embryo-toxic than average, but until there are equivalent data for other species, any conclusions about the perceived toxicity of dilbit may be confounded by species differences in sensitivity.

Spills of dilbit also do not suggest any unusual toxicity. Unpublished reports about a 2007 spill of approximately 100,000 L of dilbit to Burrard Inlet indicated that the primary acute effect was oiling of aquatic birds, most of which were rescued (Dew et al. 2015). Other effects were due to spill cleanup, i.e., removal of oiled *Fucus sp* followed by mortalities of species, such as barnacles and limpets that use *Fucus* beds as a habitat. Hydrocarbon contamination was measurable in some species of crab but not others, and contamination of sediments and mussels (*Mytilus edulis*) was still evident four years post-spill. Other long-term ecological effects were not investigated. The impacts of this spill were relatively mild, likely because it occurred in a harbour well-equipped to handle spills, the response was rapid and most dilbit was recovered. Weather and tide conditions also limited the spread of the spilled dilbit and the spill did not coincide with salmon or bird migrations. However, the sediment contamination indicated that some dilbit sank, which may account for much of the unrecovered portion.

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In fresh water, the 2010 spill of dilbit from a ruptured pipeline contaminated Talmadge Creek and the Kalamazoo River, MI, caused significant environmental impacts (reviewed in detail Chapter 8). Dilbit that flowed overland to the creek floated initially, but 10-20% sank due to the rapid evaporation of volatile components and the accumulation of soil particles and suspended particulates in the creek and river (reviewed by Dew at al. 2015). As with the Burrard Inlet spill, there were direct effects on animals in the river, including large numbers of oiled turtles, birds and mammals, most of which were treated and survived. Although there were no immediate fish kills, fish abundance and diversity decreased in oiled areas and there were clear signs of oil exposure and effects on fish, as indicated by physiological markers of exposure to petroleum hydrocarbons and poor health. Sediments contaminated with dilbit were toxic to benthic invertebrates, but toxicity varied somewhat with sediment characteristics (reviewed by Dew et al. 2015). However, the biggest impact was the damage to aquatic and riparian habitats by the spill cleanup (reviewed in Sections 4.2.7).

Overall, the direct toxic effects of dilbit in both freshwater and marine ecosystems do not seem dramatically different from those of other oils. However, dilbit does differ from other oils in its behaviour when spilled. The very rapid loss of volatiles during weathering increases viscosity and reduces the susceptibility to mechanical or chemical dispersion, which may limit the partitioning of PAHs into water (Section 4.2.3) and the extent of exposure of aquatic biota.

<u>Recommendation:</u> Research is needed to determine how the behaviour of fresh and weathered dilbit in various aquatic habitats affects exposure and toxicity to aquatic species.

More generally, the risk of toxicity of an array of oils is not absolute, but depends on viscosity and how it controls droplet formation and the partitioning of the toxic components of oil into water. Differences in the resistance to droplet formation can be reduced experimentally by chemical dispersion (Section 4.2.3) to enable comparisons of toxicity among oils with less bias from differences in viscosity. Thus, to compare toxicity among oils for ERAs, toxicity test data should be generated by methods that maximize the surface area of oil exposed to water and the bioavailability of hydrocarbons to test species (reviewed in section 4.3).

4.2.3 Chemically- vs. mechanically-dispersed oil

Chemical dispersants are used to reduce the amount of oil on the water's surface and the risks of oil to surface species, such as seabirds, marine mammals and turtles, and to shoreline marshes, estuaries and beaches. Dispersion transfers oil into the water column where it can be diluted, dissolved in water, and degraded by microbes. The downside of dispersion is the risk of toxicity from the dispersants themselves and increased exposure of aquatic species to the toxic constituents of oil (NRC 2005). Thus, chemical dispersion may increase the impacts of oil on subsurface aquatic resources by making the components more bioavailable. For example, PAHs from four different oils were six to 1,100-fold more bioavailable to juvenile trout from dispersed oil (CEWAF) than from un-dispersed (WAF) (Ramachandran et al. 2004a), depending on the oil tested. Similarly, the toxicity of CEWAF of crude, diesel and heavy fuel oils to trout embryos was greater than the toxicity of WAF when exposures were expressed as the amount of oil added to test systems (Schein et al. 2009; Adams et al. 2014a; Martin et al. 2014). *Following dispersion, the ecological risks of spilled oil will increase because the volume of water contaminated with toxic concentrations of oil may increase by up to 1,100-fold during the period of plume dilution.*

When oil is added to water, the rate of partitioning of hydrocarbons from oil to water is largely a function of droplet size, i.e. the surface-to-volume ratio (Redman 2015). With physical dispersion, oil droplet size and stability in water varies with oil characteristics such as viscosity. Heavier and highly-weathered oils are harder to disperse and appear less toxic to aquatic species. Chemical dispersants reduce the median size of oil droplets and stabilizes droplets in water and can even enhance the dispersion of heavier oils.

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Thus, the apparent increase in t oxicity following chemical dispersion can be attributed primarily to an increased bioavailability of the toxic constituents of oil.

When toxicity is expressed as oil loadings , large differences in toxicity between solutions of WAF and CEWAF have been interpreted as synergistic interaction s between the dispersant and oil toxicity (e.g., Rico-Martinez et al. 2013) . This is a common mistake when the recommended practice of measuring hydrocarbon concentrations in test solutions is not followed (NRC 2005; Coelho et al. 2013; Adams et al. 2014a). When concentrations of TPHs or TPAHs are measured in test solutions, differences in toxicity between CEWAF and WAF, among oils, and among CEWAFs prepared with different dispersants , largely disappear (e.g., NRC 2005; Wu et al. 2012; Incardona et al. 2013; Adams et al. 2014; Dussauze et al. 2015; Redman and Parkerton 2015). *Toxicity is determined by the measured amount of oil in solution, not by oil loading.*

Modern dispersants are less acutely lethal than mechanically -dispersed crude oil (WAF), and far less toxic than chemically-dispersed oil (CEWAF) (NRC 2005; Hemmer et al. 2011). Nevertheless, adding a toxic agent to disperse oil that is also toxic might seem counterintuitive.

Under ideal conditions, chemical dispersants are not readily bioavailable to aquatic species . For example, Corexit® 9500 dissolved in water is toxic to fish embryos, but non -toxic when first mixed with mineral oil to create chemically-dispersed mineral oil (Adams et al. 2014a). When properly mixed, most dispersant appeared to be sequestered by the mineral oil and not bioavailable at toxic concentrations. However, for applications under field conditions, often by airplane, there is no guarantee that all of the dispersant will land on oil sl icks or be properly mixed with oil. *Where the dispersant misses the slick or fails to mix with oil , the potential for dispersant toxicity to aquatic species in subsurface waters is increased.*

In laboratory tests of chronic embryo toxicity, the presence and effect of free dispersant are indicated by early mortality of eggs, before signs of blue sac disease appear. When mortality occurs later and coincides with the onset of blue sac disease , it is likely caused by PAHs and not by dispersant (Kuhl et al. 2013; Adams et al. 2014a). The array of gene transcriptional responses in fish embryos is also somewhat different between dispersant and dispersed oil (Olsvik et al. 2012) , suggesting a direct interaction of dispersant with transcription. *When dispersants are used at oil spills, the unique molecular responses of fish to dispersants coupled with chemical analyses of un -sequestered dispersant could provide tools to diagnose specific dispersant effects in the field.*

In older fish, dispersants may be endocrine disruptors. Nonylphenol ethoxylates, a common constituent, can degrade to nonylphenol, which interacts with the estrogen receptor of vertebrates. However, only two of eight commercially-available dispersants caused weak estrogen -receptor activity in a mammalian cell line, and no androgen receptor activity, although all dispersants were cytotoxic at high conce ntrations (Judson et al. 2010).

<u>Recommendation:</u> Research is needed to determine whether dispersants and chemically -dispersed oil are endocrine disruptors in sexually-maturing fish *in vivo*.

The intended benefit of chemical dispersion is a more rapid dilution of oil-contaminated water throughout the water column, assuming that mixing energy is sufficient. Models of the fate of WAF and CEWAF of spilled oil along 200 nautical miles of Norwegian coast over a 30 -day period showed that the overlap in distribution between toxic concentrations of oil (0.1 to 1.0 µg/L TPAH) and the eggs and larvae of northeastern Arctic cod (*Boreogadus saida*) was reduced somewhat by chemical dispersion (Vikebø et al. 2015). While the results suggest that dispersants could be used to reduce the potential for embryo toxicity, the model was highly site- and species-specific because it included many assumptions about oil and egg/embryo distribution. *Most importantly, the model did not consider the relationships among*

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concentration, exposure time, embryo developmental stage and oil toxicity. Although high concentrations of oil in water may be transitory following dispersion (i.e., less than 24 - 48 hours) (NRC 2005), only brief exposures (one to 2.4 hours) of Atlantic herring (*Clupea harengus*) eggs to realistic concentrations of CEWAF were sufficient to increase rates of deformities and mortalities at hatch and swim-up, 14 days later (McIntosh et al. 2010; Greer et al. 2012). *Thus, transient exposures of embryos to high concentrations of oil at a critically -sensitive stage of embryonic development may be highly toxic*, even though the modeling scenario indicated that 30-day average concentrations were sub-toxic.

Recommendation: Research is needed to define and model the interactions among concentration, exposure time and toxicity for different life stages of fish.

Chemical dispersion of oil also has implications for coral reef species which may be particularly sensitive because of their shallow habitats and complex species interactions. At spill sites, the exposure of coral reefs to oil will increase if chemically-dispersed oil is mixed to depths of one to 10 m. One- to five-day exposures of embryos and larvae of different coral species to CEWAF caused greater toxicity than exposures to WAF, as indicated by reduced rates of fertilization and survival (reviewed by NRC 2005). There were also long-term effects of brief exposures, expressed as elevated rates of mortality and reduced growth and coverage one to two years following a spill. Complete recovery was evident within 10 years. In lab studies, Corexit® 9500 dispersant alone was about 15 times more toxic to corals than to anemones. Sublethal exposures to dispersants or to dispersed oil reduced the feeding behavior of corals and brief exposures (8 – 24 h) to CEWAF caused delayed mortality and reduced growth, even though survivors rapidly depurated accumulated oil (Shafir et al. 2007; Mitchelmore and Baker 2010). Clearly, dispersants should not be used in the vicinity of coral reefs, a recommendation that corresponds to current practice in shallow waters.

In Canada, corals on both the Pacific and Atlantic coasts inhabit deeper and colder waters (e.g., http://ibis.geog.ubc.ca/biodiversity/efauna/BritishColumbianCorals.html) so that dispersion of surface oil is less likely to cause their exposure to petroleum hydrocarbons. However, the novel use of dispersants to manage the discharge of oil from the wellhead during the DWH oil spill demonstrated that benthic organisms in waters as deep as 1,300 m are at risk of exposure to chemically-dispersed oil.

The NRC (2005) review also questioned the assumption that oil dispersion would protect marine birds by removing oil from the surface. The protection of birds from surface oil depends on the efficiency of oil dispersion, but diving birds could be exposed to subsurface plumes of dispersed oil, and presumably this may extend to marine mammals.

Recommendation: Research is needed to: 1) assess the toxicity of dispersed oil to deepwater corals, ground fish and invertebrate species that have high economic importance (e.g., lobster, crab, scallops); 2) model the distribution of deepwater plumes of dispersed oil in relation to areas of known fisheries productivity, such as the fishing banks of Canada's east coast or unique habitats like the sponge reefs off Canada's west coast; and 3) assess the potential for effects of chemically-dispersed oil on marine bird and mammals.

Decisions on the use of dispersants following an oil spill use a 'Net Environmental Benefit Analysis' (NEBA) to identify the trade -offs between impacts on surface and subsurface resources (NRC 2005). Although the NEBA concept is logical, the effectiveness of its application is dependent the level of information available for comparisons of risk between surface versus subsurface resources and offshore versus shoreline impacts.

Recommendation: To support evidence -based decisions on net environmental benefits, baseline data are needed on abundance of species that frequent areas at risk of oil spills. Where data are inadequate, rapid methods are needed for surveying subsurface aquatic resources at the time of a

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spill. The behaviour, fate and effects of dispersed oil should be investigated under field conditions typical of Canadian waters at risk of oil spills and the toxicity of new chemical agents used for sorption, solidifying, burning, herding, specialty cleaning and bioremediation of oil (Chapter 6) assessed (Tamis et al. 2012).

4.2.4 Sediment contamination

The NRC (2005) reviewed a series of studies on experimental chemical dispersion of oil in near -shore areas of tropical, temperate and Arctic ecos ystems. While chemical dispersion reduced the amount of residual oil on shorelines, the interactions of dispersed oil with sediments were less clear, particularly with fine-grained sediments that were not the focus of most studies. A recent and very compre hensive review of the interactions among spilled oil, oil dispersants, the rate of formation of oil -mineral aggregates and oil precipitation to sediments demonstrated that dispersant use accelerates oil precipitation to sediments (Gong et al. 2014).

Bottom-dwelling (benthic) organisms will be exposed to dilbit or to oil that has settled into the sediment or is present chronically due to natural seeps. Extensive lethality has been observed when high quantities of oil reach the bottom following spills or well blowouts (Teal and Howarth 1984). Sensitive species give way to opportunistic species in such cases. Oil that has mixed with sediments can persist for very long periods of time, e.g., 30 years after the *Arrow* spill in Nova Scotia (Lee et al. 2003b) and 25 years after the EVOS in Alaska (reviewed by Ballachey et al. 2014). Persistence is due to the slow biodegradation that takes place under the anoxic conditions characteristic of benthic environments. Organisms in constant contact with contaminated sediments (infauna and epifauna) are at a higher risk of adverse impacts, including impaired feeding, growth, development and recruitment, that may alter population dynamics and community structure. In coastal waters, macroalgae (e.g, kelp) may experience decreased reproduction, bleaching and mortality upon exposure to oil, and effects on macroalgae reduce habitats and abundance of other benthic species. As discussed in Section 4.1.4.6, benthic fish species are particu larly susceptible to an increased prevalence of cancer following exposure to carcinogenic and mutagenic PAHs typical of pyrogenic PAH s, but there have been no long-term studies of the prevalence of cancer in benthic fish chronically-exposed to spilled oil.

Benthic species may also be subject to hypoxia if there is a high biochemical oxygen demand (BOD) of sediments due to organic enrichment from an oil spill and oil toxicity to benthic species could be underestimated if hypoxia is not taken into account. For example, oil from the DWH spill was more toxic to adult and embryonic sheepshead minnows (*Cyprinodon variegatus*) when tested under hypoxic conditions (2.5 mg/L dissolved oxygen) relative to normoxic conditions (>5.9 mg/L oxygen; >80% saturation) (Hedgpeth and Griffitt 2015).

<u>Recommendation:</u> Benchmarks are needed for petrogenic PAH s in marine and freshwater sediments to guide sediment cleanup following an oil spill.

4.2.5 Photo-enhanced toxicity

The aromatic components of oil (e.g., PAH) may be susceptible to photooxidation due to the interaction of UV light with double bonds (Chapter 2). The photo -modified products of these reactions include phenols or epoxides, which are more water -soluble than the parent compounds. *Thus, photooxidation of PAHs in surface waters to more water-soluble derivatives may contribute to their more rapid dilution to non-toxic concentrations. However, photo-modification may also create hydroxylated derivatives that are more toxic than the parent compound (e.g., Fallahtafti et al. 2012).*

A more important process is photo -enhanced toxicity, i.e., toxicity due to photooxidation *in vivo*, which exposes cells to highly ROS, such as singlet oxygen and hydroxy radicals. Although ROS react quickly

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and do not persist long enough in water to cause toxicity, the generation of ROS in cells allows their immediate reaction with double bonds in lipids (lipid peroxidation), proteins (inactivation of enzymes) and nucleic acids (mutations). These reactions are reproducible *in vitro*, and toxicity *in vivo* can be predicted from the physical and chemical characteristics of PAHs (Greenberg et al. 1993; Ren et al. 1994). For example, photo-enhanced embryo toxicity is greater for pyrene than for retene (Räsänen, et al. 2011). All species have oxidative defense and DNA repair systems to protect cells from damage by ROS generated by normal oxidative metabolism, but the rate of ROS production in tissues during photo-enhanced toxicity far exceeds the capacity to neutralize them. The result is a rapid destruction of cells, tissue necrosis and mortality of the exposed organism.

The risk of photo-enhanced toxicity will be greatest in the clear waters of the open ocean, in coastal and freshwater ecosystems where turbidity is low and UV light penetrates to the sediments, and at northern latitudes with seasonally long days. Species or life stages most at risk are transparent or semi-transparent (Barron and Ka'aihue 2001), including many zooplankton species (sub-Arctic copepods) (Duesterloh et al. 2002), crab larvae (Alloy et al. 2015) and the embryos of fish that are un-pigmented at hatch (e.g., vendace [Coregonus albula]) (Vehniainen et al. 2003). There is little risk to pigmented embryos (e.g., pike; Esox Lucius) (Hakkinen et al. 2004; Räsänen et al. 2011) or to juvenile and adult fish (Barron et al. 2005).

For marine species, such as Pacific herring, co-exposure to UV light, including normal sunlight, increased the toxicity of WAF and CEWAF of crude oil by up to 48-fold (Barron et al. 2003; NRC 2005), as was the case for freshwater zebrafish embryos exposed to crude oil (ANS, MC-252) and to HFO (Hatlen et al. 2010; Incardona et al. 2012b). WAF alone increased the prevalence of chronic, sublethal cardiotoxicity in exposed embryos, whereas WAF combined with natural sunlight caused more rapid mortality due to severe tissue necrosis. Cardiotoxicity was correlated to concentrations of tricyclic PAHs in test oils, but photo-enhanced toxicity may involve other compounds, such as heterocycles (e.g., carbazoles). UV light caused a greater increase in toxicity of HFO compared to ANS (Hatlen et al. 2010), likely because of higher concentrations of TPAHs involved in photo-transformations.

While there are many studies of photo-enhanced toxicity to fish and aquatic invertebrates, the significance remains controversial because few field studies have confirmed these effects at actual oil spills. Sellin Jeffries et al. (2013) developed a risk model relating PAH concentrations in surface waters following the 1989 EVOS to the depth of UV-A light penetration and the occurrence of embryos of Pacific herring in surface waters. They concluded that less than 1% of the herring embryo biomass would have been present in surface waters containing sufficient PAHs to cause injury by photo-enhanced toxicity. In contrast, field experiments with caged Pacific herring embryos and passive water samplers suggested that the toxicity of HFO spilled by the grounding of the *Cosco-Busan* in San Francisco Bay in November 2007 was enhanced by natural UV light (Incardona et al. 2011). The passive samplers demonstrated the presence of other contaminants derived from urbanized watersheds at both oiled and reference sites. Although hydrocarbons associated with photo-enhanced toxicity were only present at oiled sites, the results do not preclude interactive toxicity among petroleum hydrocarbons, anthropogenic compounds and natural stressors, such as temperatures that fluctuate with tidal cycles.

<u>Recommendation:</u> Additional studies are needed to verify that photo-enhanced toxicity occurs at oil spills, and the extent to which it may interact with other anthropogenic contaminants and natural stressors.

4.2.6 Arctic vs. temperate spills

The interest in oil resources by Arctic coastal nations and the expansion of oil exploration and development along the Mackenzie River Valley and in the Beaufort Sea, raise concerns about the impacts of oil spills in sub-Arctic and Arctic ecosystems, and the toxicity of oil at low temperatures. Recent

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reviews of oil spills under Arctic conditions conclude that impacts will depend on the interactions among the unique physical characteristics of the environment (e.g., the presence of sea ice), the chemistry, fate and behaviour of spilled oil at low temperatures (Chapters 2, 3 and 6), and the biology of primary, secondary and tertiary production by species uniquely adapted to Arctic conditions (Lee et al. 2011; AOSRT 2014).

The perceived risk of oil toxicity to Arctic species is derived primarily from laboratory toxicity studies, most focused on fresh oil. Most field studies of spilled oil in the Arctic, concern Arctic waters of Norway (AOSRT 2014), Chapter 8 and Lee et al. (2011) review the Baffin Island Oil Spill experiment (BIOS) in detail. The AOSRT (2014) review was concerned primarily with marine conditions, and effects on freshwater species were not addressed. However, as indicated in Section 4.2.1, there appears to be little systematic difference between freshwater and marine species in sensitivity to oil exposure, and presumably this extends to Arctic species, although no studies were found that tested this assumption.

The implications for Arctic mammals are covered briefly in Section 4.1.7 and relate mostly to the increased potential for exposure to oil when it is confined to breathing holes in ice. For Arctic invertebrates and fish, effects will depend on the extent of their exposure to oil entrained or dissolved in water or trapped under ice where species congregate. Under-ice algae that grow in light passing through surface ice, support communities of organisms, including periphyton grazers, predatory fish and marine mammals. These under-ice communities would be uniquely sensitive to a subsurface discharge of oil because buoyant oil would accumulate in high concentrations under the ice, where there may be a risk of photo-enhanced toxicity (Lee et al. 2011; Section 4.2.5). As AOSRT (2014) concluded, the exposure of marine Arctic species to oil could correspond to two scenarios: spiked exposures (brief and rapidly declining exposure; see Section 4.3.1) typical of open ocean spills; and prolonged or chronic exposures associated with oil trapped under ice.

Other unique conditions in the coastal shelf of the Beaufort Sea that could enhance the impacts of oil spills have been identified (Carmack and Macdonald, 2002; Lee et al. 2011; ADNR 2014). For example, a thermal bar formed each summer along the shore west of the MacKenzie River creates a zone of low salinity water where river and salt water mix. This zone is highly productive and attracts salinity-tolerant fish species from coastal tributaries. Oil contamination in this zone could have significant impacts on abundance and productivity of both freshwater and marine fish species, and other species such as beluga whales that seasonally occupy critical habitat in the near-shore.

Whether Arctic species are uniquely sensitive to oil due to adaptations to life between -1 and 2 °C remains an open question. Comparisons of toxicity among temperatures will be complicated by the effects of temperature on the kinetics of partitioning of hydrocarbons from oil to water, the pharmacokinetics of bioaccumulation, and the activity of enzymes involved in metabolism and excretion of hydrocarbons. In assessing the risks to fish, consideration must be given to their unique adaptations to Arctic conditions, such as high tissue concentrations of antifreeze proteins. For example, the unusually high sensitivity of Arctic cod to oil exposure may be related to a lack of kidney glomeruli to limit the loss of antifreeze proteins. Excretion of PAH metabolites via the bile instead of the kidneys may prolong their exposure to PAHs by reducing excretion rates. Because *comparative data on hydrocarbon metabolism and toxicity are sparse, tools developed for monitoring oil exposure and effects in temperate species should be interpreted with caution when assessing the potential impacts of oil on Arctic fish species.*

Preliminary studies suggest little difference in oil toxicity between Arctic and temperate species (AOSRT 2014). A comparison of published data on the toxicity of two components of crude oil and 13 crude oils to an array of marine species across five phyla (annelids, arthropods, chordates, echinoderms and molluscs) concluded that there was no systematic difference in acute lethality (<8 day exposures) between temperate and Arctic species (de Hoop et al. 2011). Similarly, biomarker responses of Arctic cod exposed to the water soluble fraction of crude oil (e.g. induction of cytochrome P450 and antioxidant enzymes;

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DNA damage) were identical in direction and fell within the range of those observed for temperate species exposed to similar concentrations of oil (Nahrgang et al. 2010). *de Hoop et al. (2011) concluded that data from temperate species could be used for a first approximation of risks of oil spills to Arctic ecosystems.* However, chemically-dispersed oil was less toxic at 2 °C to Arctic species of fish and invertebrates than un -dispersed (Gardiner et al. 2013), in contrast to greater toxicity of CEWAF to temperate species at 10 -28 °C (Section 4.2.3). The greater toxicity of WAF than CEWAF at low temperatures may be explained by a more prolonged retention of acutely toxic volatile compounds at cold temperatures compared to warm (Perkins et al. 2005). Understanding the effect of temperature on the relative toxicity of chemically -dispersed oil will require detailed chemical analyses of hydrocarbons in test solutions and tissues of test organisms.

Overall, the differences in oil spill effects between Arctic and temperate regions appear to be driven more by factors that affect exposure of organisms to oil than by differences in toxicity of oil. Recent reviews of oil toxicity in Arctic ecosystems by Olsen et al. (2013) and AOSRT (2014) identified research needs similar to those of temperate ecosystems.

Recommendation: Research on the effects of oil should take into consideration unique aspects of Arctic ecosystems by considering: 1) interactions among low temperature, physiological adaptations of Arctic species to low temperature and the kinetics of hydrocarbon solubility, uptake and excretion; 2) the habitat, life histories and reproductive capaci ty of regional species at risk ; 3) chronic or sublethal endpoints related to growth and embryonic development; 4) heavy fuel oils widely used in Arctic shipping; 5) the efficacy and effects of oil dispersion under Arctic conditions; 6) long-term studies of ecosystem recovery; and 7) inclusion of Arctic communities in the d esign and conduct of research.

4.2.7 Marine vs. freshwater spills

The history of oil spills is dominated by major shipping accidents at sea or near marine coastal ecosystems. Much of what is known about the potential effects of oil spills is derived from detailed research following incidents, such as the EVOS in Prince William Sound, AK, in 1989, and the more recent DWH spill in the Gulf of Mexico in 2010. However, as reviewed in Chapter 1, the frequency of marine shipping accidents has been declining due to improvements in tanker safety programs and construction standards. In Canada, most recent spills are inland, reflecting the locations where oil is produced, transported and used.

As the result of an aging pipeline infrastructure, major oil spills have occurred in Russia, such as the Komi spill that resulted in the release of 100,000 -350,000 tonnes of crude oil and produced water into a river near the town of Usinsk (Owens et al. 1999) . As media attention and scientific reporting on these spills have been limited, the public's attention on oil spills in North America has been focused on marine oil spills and the large cleanup efforts involved. It is important to exam ine the level of environmental impacts, costs and technical challenges that may be encountered following spills to fresh water.

A comparison of the 2010 DWH oil spill from a well blow-out in the Gulf of Mexico and the 2010 spill of Western Canadian Select dilbit (Table 2.2) to the Kalamazoo River, MI, from a pipeline break highlights some important differences between marine and freshwater spills (Hodson and Williams 2012 ; Chapter 8.7). The most obvious difference was one of scale. The DWH spill discharged about 780,000 m^3 of crude oil over a 12 -week period, contaminating an area ranging from 6,500 to 180,000 km^2. By contrast the Kalamazoo River spill lasted only17 hours, the volume lost was 235-fold smaller (3,320 m^3), and only 60 km of river was oiled. At first glance, the potential for ecological impacts would appear much smaller for the Kalamazoo River. However, in both cases, there were significant impacts on aquatic and shoreline ecosystems, significant technical challenges in dispersing or recovering oil, and the cleanup continued for at least four years due to residual oil present as slicks, tar balls and contaminated sediments.

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The challenges of cleaning up a freshwater spill were reflected in a substantial difference in the estimated costs of cleanup ($50 US/L spilled for the DWH vs. $300/L spilled for the Kalamazoo River) (Hodson and Williams 2012) . The higher 'per litre' costs to clean up the Kalamazoo River spill seems counterintuitive given the difference in size the affected ecosystems. In marine systems, tidal cycles nourish broad shoreline and intertidal ecosystem s that are rich in species adapted to daily flooding, exposure to air and input of nutrients. Because of tides, the intertidal area subject to oil contamination, and the number of species affected, is quite large. In contrast, water level fluctuations in freshwater systems are much smaller and less frequent, the shoreline ecosystem is narrower and the shoreline species diversity and density are lower (Sergy and Owens 2011).

Relative to the Gulf of Mexico, the Kalamazoo River is of low volume with a limited dilution capacity and a limited area for oil to spread and d isperse. Dilbit remained in thick layers with less opportunity for weathering by photolysis and biodegradation and a greater opportunity for exposure of aquatic biota to toxic hydrocarbons. Although mixing energy from wind or waves was relatively low in the Kalamazoo River, water depths range d from only a few centimeters to a few meters, increasing the interactions between surface water and sediments. The high ratio of shoreline length-to-water surface area meant that a greater proportion of shoreline was contaminated and seasonal flooding carried oil into forested riparian lands.

Although the DWH spill was much larger than the Kalamazoo River spill, the affected Gulf of Mexico open water ecosystem was proportionately larger (Cleveland 2013), ensuring a very high dilution capacity and opportunity for oil to spread, disperse and degrade. The open sea provided abundant mixing energy due to wind and waves and an extended time before oil reached shore (four to five weeks)(Cleveland 2013), where impacts were similar in nature to those of the Kalamazoo River. Although the great depth of the Gulf (1,500 m at the site of the well blowout) limited interactions between surface oil and sediments, a unique feature of the DWH spill was the physical and chemical dispe rsion of oil at the wellhead. Approximately, 75% of discharged oil did not reach the surface, forming a subsurface plume at 1,000 - 1,400 m depth. This plume appears to have contaminated sediments of the Gulf of Mexico over a wide area (Valentine et al. 2014), but the extent and degree of contamination is still a matter of debate and research. In the Kalamazoo River, contamination of shallow (< 10 m) sediments was evident as frequent sheens that resurfaced from the sediments to the relatively calm water surface and considerable effort was expended in trying to recover sunken oil.

In offshore marine waters, oil usually floats because its specific gravity is less than that of salt water (Chapter 2) and low turbidity reduces the likelihood of oil -mineral aggregat es that may sink and contaminate sediments. As floating oil spreads, surface slicks thin and weather rapidly, and are subject to more rapid photolysis and biodegradation (Chapter 2). For the DWH spill, winds and currents kept much of the surface oil offshore, enabling recovery by skimming, burning of surface oil and surface chemical dispersion for enhanced dilution and biodegradation. Estimates of the fate of the spilled oil are highly uncertain, but it is likely that only a relatively small proportion (but still a large volume) of the oil reached shore, although weathered oil continues to wash ashore as tar balls or oily materials. For the Kalamazoo River spill, the specific gravity of un-weathered dilbit was close to that of fresh water, increasing the risk of sinking as it weathered and lost its lighter components. The risk of dilbit sinking was increased by the formation of oil-mineral aggregates (Figure 2.2) due to contact with soil before entering the Kalamazoo River and with high concentrations of suspended solids in the water caused by flooding.

Effects of oil on aquatic receptors vary with their relative sensitivity and extent of exposure. No studies have been published that specifically compare the relative sensitivity of marine and freshwater species, although publications on toxicity (**Figure 4.2** , Appendix D, Table D.1) do not suggest any major differences. However, the bioavailability of PAH s to salinity-tolerant fish species exposed to WAF and CEWAF decreased with increasing salinity (Ramachandran et al. 2006), suggesting a greater sensitivity of fish to spilled oil in fresh or low -salinity waters. In marine ecosystems, the re is a potential to harm

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more species than in fresh water due to a higher biodiversity. In contrast the potential impacts of spills on single species may be more severe in freshwater habitats. Avoidance is possible (e.g., in clean tributaries), but habitats are small and physically constrained and there is less oil dilution. *Overall, oil spill effects will be site- and species-specific, depending on how the biology and ecology of each species interact with the potential for oil exposure. In both marine and freshwater ecosystems, impacts will vary with seasonal aggregations of species for spawning, synchronized hatching of eggs, schooling behaviour and habitat selection.*

Oil spill cleanup options (Chapter 7), *such as booming and skimming, burning of surface oil and chemical dispersion, are well established for marine ecosystems and are designed for open water applications. For offshore spills, delays may be significant before oil is blown onto sensitive shorelines, allowing the opportunity to plan and prepare a response to protect those shorelines.* For the DWH spill, the deepwater plume of dispersed oil presented new challenges in understanding potential impacts and how to mitigate them. Although the plume was 'good news' in that the oil was apparently 'gone', subsequent studies identified a 'bathtub ring' of contamination at depths where the plume encountered sediments (i.e., at 1,500 m) (Valentine et al. 2014). The significance of this contamination and the extent of damage to benthic ecosystems is still a subject of active research.

At open ocean spills, cleanup efforts are viewed as mostly positive because oil is physically removed from the surface (skimming, burning, chemical dispersion) to protect surface species and sensitive shorelines. The large-scale of the open ocean also reduces the risk and overall impacts of smoke from burning oil and the transient increase in dispersed oil concentrations in subsurface waters. In freshwater ecosystems, burning and chemical dispersion of oil are not accepted practices because they present toxicity risks to human populations and to small aquatic ecosystems. While booming and skimming work in some circumstances, access of equipment to oil spills may be limited by topography (e.g., canyons), distance (remote wilderness with no road access) or season (spring thawing of ice roads). Strong currents in high-gradient rivers also reduce the effectiveness of booms and skimmers, particularly during spring floods when ice and debris are present. The sinking of dilbit in the Kalamazoo River presented challenges for cleanup technologies and a heightened standard of 'How clean is clean enough?' due to its proximity to rural and urban development. New technologies were developed to capture oil droplets suspended in the water column ('snares') and to refloat oil from contaminated sediments ('poling', 'tillers', water jets, etc.), but their overall effectiveness in terms of cost and percentage of oil recovered are not well established (Chapter 7).

Spill cleanup often causes significant habitat damage, in addition to the physical and chemical effects of oil. The effects of mechanical cleanup of wetlands and marshes along shorelines differ between marine and freshwater ecosystems. Root systems in marine marshes are dense and can bear greater weight than more porous root systems of freshwater marshes (Sergy and Owens 2011), so greater care is needed in freshwater ecosystems to minimize habitat damage. Shorelines and riparian lands are damaged by the construction of access roads and stable pads for large equipment, by removal of oiled vegetation and un-oiled vegetation to access shorelines, the excavation of contaminated soils and the destabilization and erosion of shorelines, marshlands and river channels following excavation. For the Kalamazoo River watershed, riparian lands along both sides of Talmadge Creek (site of the spill) were deforested for access roads and some islands that had been over-washed by oil during flooding were removed completely. Habitat destruction on this scale requires extensive remediation and the impacts on ecosystem structure, function and services (e.g., access by locals for recreation) may be evident for decades. Other examples are reviewed in Chapter 8.

Overall, the scale of oil spills in freshwater ecosystems may be smaller than in marine systems, but the severity of impacts relative to the amount of oil spilled may be greater due to the smaller scope for dilution and degradation of oil, the close proximity of shorelines and sediments, the confinement of aquatic species to the spill sites and high densities of single species during migration or spawning. The

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fate of much of the oil spilled to freshwater ecosystems is often unknown, which raises concerns for potential interactions of surface oil with sediments. Entrainment of droplet oil into bed sediments by surface water (hyporheic flows) creates a risk of groundwater contamination and toxicity to early life stages of aquatic species that live in or spawn in sediments. The technical difficulties of cleanup are aggravated by limitations on the use of common tools, such as chemical dispersants and burning, the inaccessibility of some spill sites and the potential for greater damage to sensitive ecosystems than is caused by the oil itself. The pressures on companies that spilled oil, on agencies responsible for regulation and on responders cleaning up the oil to justify their actions, to clean up the spill quickly and to prevent future spills may be that much greater when spills are near urban development.

Recommendation: Research is needed on the behaviour and effects of oil in freshwater ecosystems to take into account the diverse scales, processes, water movements and biological communities in such systems.

4.2.8 Interactive and cumulative effects

The effects of oil spills on aquatic ecosystems are often viewed in isolation of other stressors, particularly if they occur in the open ocean. However, with shipment of oil through Canada's major ports, waterways and watersheds, and spills from pipelines in areas of urban and industrial development, there is a potential for oil spills to waters already affected by contaminants from other activities or legacy contaminated sites. A few examples include:

- Mining effluents: metals, metalloids (e.g. selenium), mine flotation reagents, acid, suspended solids;
- Pulp mill effluents: organic contaminants, chemical and biochemical oxygen demand (COD and BOD);
- Municipal effluents: pharmaceuticals, personal care products, BOD, nutrients;
- Agriculture: pesticides, nutrients, suspended solids, BOD, pathogens (livestock); and
- Wood preservative plants: creosote, copper, chromium, arsenic.

In addition to the potential for interactive toxicity, organic wastes can alter oil fate and behaviour. High concentrations of dissolved organic material (e.g., pulp mill and municipal effluents) may act as chemical dispersants, promoting the formation of oil droplets and the solubilization of petroleum hydrocarbons. As with chemical dispersants, dispersion by organic wastes may remove surface slicks more rapidly than in pristine waters, diluting oil to non-toxic concentrations and promoting biodegradation. However, in confined and shallow fresh waters, they would also increase the exposure of aquatic species to toxic concentrations of oil without the benefit of open water dilution. Similarly, increased turbidity due to soil erosion (agriculture, mining, forestry) and the discharge of organic particulates in effluents may promote the formation of oil-mineral aggregates and increase contamination of sediments.

Other stressors could include normal events in the life cycle aquatic species. A massive spring die-off of bluegill sunfish (*Lepomis macrochirus*) in Lake Winona, MN, followed a winter spill of Bunker C oil (Fremling 1981). Mortality was delayed until spring spawning, likely due to the combined stresses from oil exposure, loss of condition over the winter, rising temperatures and spawning.

Natural oil seeps or contaminated sediments that occur in oil-rich areas such as the McMurray oil sands formation in the Athabasca River watershed, and methane seeps in the Arctic Ocean may also influence the fate and effects of spilled oil. Natural sources of hydrocarbons create a potential for background or baseline toxicity (e.g., Colavecchia et al. 2004, 2006) and foster microbial communities adapted to biodegradation of oil. The balance between an increase in the 'toxic load' of hydrocarbons and the rate of their removal by microbial degradation under different environmental conditions will be site-specific.

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In summary, the cumulative and interactive effects of oil and other contaminants are not well defined and are little studied.

<u>Recommendation:</u> Research is needed to determine interactions between other anthropogenic contaminants and the fate, behaviour and effects of oil in coastal and freshwater ecosystems, and to develop methods that discriminate between the effects of oil and other stressors.

4.3 Limitations of Current Toxicity Test Methods with Oil

The quality of ERAs for proposed petroleum developments and EIAs following oil spills depend entirely on the quality of data used to predict impacts and to incorporate into models. Toxicity data are derived primarily from laboratory tests that describe the quantitative r elationships between oil concentrations and the responses of exposed organisms. Ideally, toxicity data should be comparable among laboratories. However, as discussed in Section 4.1.4.1.1, the accuracy and relevance of toxicity benchmarks derived from toxic ity data for a diverse array of species developed by a variety of laboratories depend on a number of largely untested assumptions, including that:

- *Concentrations of petroleum hydrocarbons are known and constant during a toxicity test;*
- *The relationship betw een the amount of oil added to a test solution and the concentration of petroleum hydrocarbons in solution is monotonic;*
- *All petroleum hydrocarbons are freely dissolved in aqueous test solutions;*
- *The constituents of oil as a complex mixture remain in const ant proportion to each other across the range of test concentrations and over time; and*
- *The exposure scenarios in laboratory tests match the scenarios of actual oil spills.*

For most published studies of oil toxicity, these assumptions are rarely tested, an d when the assumptions have been tested, they are rarely met due to the low water solubility of hydrocarbons . Redman and Parkerton (2015) reviewed the implications for the outcome of toxicity tests with crude oil, and provided recommendations for the optimal test method, cited throughout the following text. *This section provides strong support for improving and standardizing methods for oil toxicity testing and chemical analyses of test solutions. Standardized tests provide a technically sound basis for comparisons of toxicity among species, environmental condition, and different sources of oil, but they do not necessarily provide data that match scenarios of actual spills.*

4.3.1 Constant vs. time-varying exposure concentrations

Because the components of oil are hydrophobic, with log K_{ow} values ranging from <3 to >6.5, the loss of hydrocarbons from test solutions is continuous, and test concentrations are not constant. The more water-soluble LMW components are lost primarily by volatilization and possibly by biodegradation. The less soluble components are lost by adsorption to the walls of exposure vessels, to the test organisms and to surface films. As a consequence, in static toxicity tests without daily renewal of test solutions, there is a continuous decline in concentrations of oil over the course of the experiment. Within 24 to 48 hours, virtually no hydrocarbons are bioavailable to test species (e.g., alkyl PAH) (Kiparissis et al. 2003). These exposures are equivalent to 'spiked exposures' (NRC 2005), the continuous dilution method recommended by CROSERF (*Chemical Response to Oil Spills Ecological Effects Research Forum* ; see below).

In tests of solutions of hydrocarbons that are renewed daily (static daily -renewal protocol), the loss of hydrocarbons from solution creates a 24 -hour saw -tooth pattern of exposure (Kiparissis et al. 2003). These test conditions are the most commonly reported for oil toxicity tests. However, in interpreting these tests to develop benchmarks, it is unknown if test organisms respond to the nominal concentration

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(amount added to the test solution), the initial concentration (time = 0 hour), the average concentration over the duration of the test, or some integral of concentration and time (e.g., Toxicity Index (TI) = concentration x time) (NRC 2005). *The issue is further complicated because the rate of loss of hydrophobic compounds from test solutions increases with hydrophobicity (K_{ow}), creating a mixture in which the relative proportions of each constituent are changing constantly. These test conditions create a great deal of uncertainty in risk assessments of oil toxicity, and in comparing toxicity among different oils, different species and different environmental conditions.*

These problems can be resolved, in part, with continuous-flow exposure systems. These include: oil-desorption columns (partitioning of hydrocarbons from oil-coated gravel to flowing interstitial water) (Carls et al. 1999; Martin et al. 2014); re-circulating tanks with pumps to recycle and mix surface oil into the water of exposure tanks (e.g., Dussauze et al. 2015); and continuous addition of fresh oil to flowing water, e.g., using jets to mechanically generate reproducible suspensions of oil droplets in exposure solutions (Nordtug et al. 2011a). Relative to static or daily-renewal protocols, oil-desorption columns slow the rate of decline in oil concentrations. However, they cannot sustain constant concentrations because oil-water partitioning depletes the finite amount of hydrocarbons in the column. The continuous re-circulation of surface oil in toxicity test tanks sustains relatively constant concentrations of hydrocarbons in water. However, concentrations decline somewhat (Dussauze et al. 2015) due to weathering or absorption to the apparatus. The most stable exposure system is the continuous generation of fresh suspensions of oil droplets by oil-water injection (Nordtug et al. 2011a). In tests with haddock embryos, this system provided a relatively constant exposure regime in terms of TPHs and TPAHs, with less than a 25% decline in concentrations over an 18-day test (Sørhus et al. 2015). The primary drawback is the need to dispose of a large amount of oily wastewater. *Although methods for generating a continuous flow of oil-contaminated water are available, they are not yet widely used or characterized under different salinities, temperatures and flow conditions.*

<u>Recommendation:</u> Research is needed to test and validate continuous-flow methods that improve the stability of exposure concentrations throughout oil toxicity tests.

An alternative way to interpret and apply oil toxicity data developed with time-varying exposures is to incorporate pharmacokinetic equations describing rates of hydrocarbon accumulation, metabolism and depuration into models that relate hydrophobicity to tissue doses causing toxicity (Redman and Parkerton 2015) (Section 4.1.4.1). Mathews et al. (2008) tested this approach with the target lipid model (TLM) developed by diToro et al. (2000). Although the results were promising, data for pharmacokinetics of individual hydrocarbons were scarce. Although the potential role of hydrocarbon metabolism by cytochrome P450 enzymes (sections 4.1.4.4 to 4.1.4.6) was acknowledged, metabolism and the toxicity of metabolites were not included in the model. The TLM modified for pharmacokinetics included the same untested assumptions about uptake, MoA and toxicity of hydrocarbons as the original (Sections 4.1.4.1 and 4.1.4.2), and the data sets used to develop and validate the model were not designed for that purpose. The modified model also did not address the effects of environmental (e.g., temperature, salinity) and biological factors (e.g., species) on rates of PAH uptake, metabolism and depuration.

<u>Recommendation:</u> The uptake and depuration rate constants for a wide array of individual PAHs and alkyl PAHs should be measured for a variety of species and life stages of fish, and at different temperatures and salinities.

4.3.2 Exposure time

Acute toxicity tests typically involve 24 to 96 hour exposures and chronic tests may last days, weeks or months, depending on the response being measured and the life history of the test organism. These tests were originally developed to regulate specific chemicals and the discharge of industrial effluents and may

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not represent exposure conditions at actual oil spills. NRC (2005) criticized 48 to 96 hour tests because they would be overly conservative, predicting greater toxicity than would be observed in an oil spill where the acutely lethal LMW components would volatilize or be biodegraded in less than 24 hours. However, this critique was written before the DWH oil spill for which there was a continuous discharge of oil for four months, broadening spill scenarios to include longer exposure times.

Nevertheless, *most oil spills are of short duration and information is needed on the long-term effects of short-term exposures, i.e., delayed or lingering toxicity.* Studies of delayed effects are few in number, although recent studies demonstrated that brief exposures (1 to 2.4 hours) of Atlantic herring embryos to realistic concentrations of CEWAF of crude oil following egg fertilization caused equivalent effects on embryo deformities and survival as continuous exposures throughout embryo development (Section 4.2.3). Exposures of fish embryos to low concentrations of oil impaired swimming capacity when the fish had developed as juveniles, well after the exposure had ceased and despite a lack of obvious toxicity during the embryo exposure (Section 4.1.4.5). Longer-term delayed effects on pink salmon migration were also evident years after embryos survived exposure to WAF of ANS crude (Section 4.1.1.4).

If exposures of fish embryos for less than 48 hours are sufficient to impair recruitment, growth, survival and reproduction of juveniles, the impacts of brief oil exposures may be far greater than anticipated.

<u>Recommendation:</u> Research is needed to develop brief and simplified toxicity testing protocols calibrated to realistic exposure scenarios and that assess delayed effects on fish embryos of acute exposures to oil.

4.3.3 Measurements of oil concentrations in toxicity tests

Varying proportions and concentrations of hydrocarbons in test solutions during toxicity tests generate considerable uncertainty in the estimated toxicity of oil. Uncertainty is even greater if hydrocarbon concentrations in test solutions are not analyzed. Toxicity expressed in terms of oil loadings greatly underestimates actual toxicity. Without a detailed chemical analysis of test solutions, the effects of chemical dispersion on oil toxicity can also be misinterpreted (Section 4.2.3).

Bias in estimates of toxicity of mechanically- and chemically-dispersed oil can be reduced by frequent analyses of test solutions to define the actual gradient of exposure concentrations, to assess whether exposures are constant throughout the test, and to establish means and variances for calculating endpoints (e.g., EC50s). Frequent analyses of test solutions by GC-MS provide a detailed understanding of test concentrations of oil and of the hydrocarbons associated with toxicity. Unfortunately, they can also be very costly, which may explain why many papers do not include measured concentrations of oil. Alternatives include fluorescence spectroscopy that can be calibrated against analyses of TPHs and TPAHs (Adams et al. 2014b; Martin et al. 2014). Fluorimetry is well-established as a survey tool in field studies of oil spills (e.g., Kim et al. 2010; Lee et al. 2012). At the concentrations of oil typical of chronic toxicity, there is a linear correlation between measures of TPAHs by GC-MS and concentrations of TPHs determined by fluorescence, enabling the expression of endpoints in terms of TPAHs (Martin et al. 2014), provided that quality assurance and quality control methods are followed.

As a result of the widespread use of dispersant during the DWH spill, there is also a need for inexpensive methods to characterize the concentrations of the ingredients of dispersants, in toxicity test solutions and treated surface waters (Place et al. 2010). Dispersant analyses would clarify the relative contributions of dispersant and oil to the observed toxicity of dispersed oil when dispersants are not completely sequestered by oil droplets. The active ingredient of Corexit® 9500 (dioctyl sodium sulfosuccinate) has been measured successfully by liquid chromatography-mass spectrometry (Kujawinski et al. 2011), but the cost precludes its frequent application for most toxicology research.

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<u>Recommendation:</u> Inexpensive analytical methods should be developed for petroleum hydrocarbons and for chemical dispersants that can detect th e toxic components at concentrations at or below the threshold of toxicity for most species (Section 4.3).

4.3.4 Confounding of risk assessments by oil droplets

The presence of microscopic oil droplets (<100 μm) in test solutions introduces another major source of uncertainty in estimate s of oil toxicity. Uncertainties are due to significant differences among laboratories in protocols for preparing solutions of oil for toxicity tests, in design of exposure systems and in the analytical characterization of test so lutions (Redman and Parkerton 2015). Chemical analyses of test solutions typically combine both particulate and dissolved oil because water samples are extracted with solvent. Each measurement represents the sum of hydrocarbons in droplet and dissolved pha ses, but results are often interpreted as if all hydrocarbons were freely dissolved and bioavailable. Th is bias applies equally to analyses of water sampled during oil spills. Thus, risk assessments comparing measured toxicity to measure d concentrations of hydrocarbons in surface waters will be confounded by significant overestimates of oil concentrations in water and underestimates of oil toxicity. The clear implication for risk and impact assessments is that reliance on published toxicity data, without consideration of the inherent biases in toxicity publications, creates a high risk of predicting false positives (effects when none would occur) or false negatives (no effects when they would occur). These biases reduce the extent to which data can be compar ed or combined across studies and limit the database for assessments to a very small number of studies.

For many fish species, embryo toxicity appears to be caused by the dissolved components of oil, not particulate or droplet oil (e.g., Carls et al. 2008; Olsvik et al. 2011). If true, *the role of droplets is simply to transfer free -phase oil to the subsurface where partitioning from droplets to water contributes dissolved hydrocarbons (Redman 2015). However, droplets may also be a direct route of oil exposure for some species but not others* . Oil droplets have been observed to adhere to the gills of juvenile trout exposed to chemically-dispersed oil in fresh water (Ramachandran et al. 2004b) and to the eggs of Atlantic haddock in salt water, but not to eggs of Atlantic cod (Sørhus et al. 2015). Oil was more embryotoxic to haddock than to cod (Nordtug et al. 2011b; Sørhus et al. 2015), suggesting little contribution of droplets to toxicity. These results were consistent with experiments in which filtered (no droplets) and unfiltered (droplets) of mechanically-dispersed oil caused little difference in the level of gene expression of cod larvae (Olsvik et al. 2011). Relative to WAF, CEWAF of North Sea oil enhanced the accumulation and toxicity of oil droplet s to filter-feeding invertebrates (*Calanus finnmarchicus*) (Nordtug et al. 2015). Other filter feeders (e.g., mussels) accumulate both dissolved and particulate oil but the relative contribution of each phase is not well-defined and may vary among species.

<u>Recommendation:</u> Research is needed to determine the extent to which oil droplets contribute to oil exposure, why droplets interact directly with some species but not others , and the mechanisms and environmental factors that promote droplet interactions with biological membranes.

There are no commonly-accepted methods for separating droplets from dissolved oil in samples collected, although centrifugation and filtration have been tried . Centrifugation is a challenge due to the small differences in specifi c gravity between water and oil . Contact of unpreserved samples with centrifuge tubes also risks the loss of dissolved hydrocarbons by absorption to the tube. Similarly, concentrations of dissolved hydrocarbons in water samples may be increased or decreased during filtration because droplets that accumulate on filters create an increasing volume of liquid phase oil for oil -water partitioning. To avoid these problems, hydrocarbons in water samples could be extracted and concentrated with passive samplers instead of solvents (Redman 2015). With solid phase micro extraction, dissolved hydrocarbons are absorbed onto a solid matrix followed by GC-MS analysis of the solid phase (Parkerton et al. 2000; de Perre et al. 2014). Solid phase micro extraction reduces the cost, time and potential error of oil analyses,

but assumes that the sorbents accumulate hydrocarbons only from the dissolved fraction in water and not from droplet oil.

Droplet bias can be detected by measuring HMW markers of oil, such as phytane, which relatively water insoluble and should be non -detectable if solutions contain only dissolved hydrocarbons. The extent of droplet contamination can be estimated from the ratios of TPAH to HMW alkane concentrations in whole oil and in oil solutions, or by com paring measured concentrations of PAH to their solubility limits calculated from K_{ow} (Redman et al. 2012, 2014). At the moment, few clear strategies exist in fluorescence spectrometry for discriminating dissolved from particulate oil in field samples, although measurements of fluorescence quenching have been used to estimate the interferences by dissolved organic mat ter in laboratory-prepared mixtures (de Perre et al. 2014).

<u>Recommendation:</u> Research is needed to optimize the use of solid phase micro extractions for measuring dissolved hydrocarbons, to separate dissolved from particulate oil in fluorescence analyses, and to harmonize sampling and analytical methods between field and laboratory studies.

Landrum et al. (2012) reviewed the challenges of testing the toxicity of complex mixtures of organic chemicals and arrived at similar conclusions, with several useful rec ommendations to improve the utility of experimental data in ERAs (Text box). While it may not be practical to apply all of these recommendations to oil toxicity tests, they should be considered when interpreting and repor ting results of toxicity tests.

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Guidance for evaluating the aquatic toxicity of complex mixtures of organic chemicals (Landrum et al. 2012)

Establishing a dose-response relationship:

- Separate all dose-response analyses by endpoint;
- Differentiate constant exposures from pulsed or intermittent exposures;
- Identify potential biotransformation products that may be ultimately responsible for the observed effect(s);
- Measure both external concentration and internal (body residue) dose for mixture components, including biotransformation products, to determine those at which different toxicity endpoints are observed and to help identify potential biotransformation products that may be responsible for the observed effect(s);
- If possible, measure body residues for all chemicals in the mixture suspected of contributing to mixture toxicity; this provides more certainty than measures of external concentrations for actual target site concentrations, particularly when there are multiple exposure routes;
- If all mixture components are not or cannot be analyzed, transparently document the associated uncertainties so that surety of cause is not assumed;
- Ensure that measurement endpoints are appropriate to the mechanisms of toxic action of the mixture and its components so that comparisons are not 'apples to oranges'. Consider and account for toxicity modifying factors that can confound apparent conclusions, including environmental conditions of the test, life stages and behaviour of the test organisms, possible external or internal biotransformation of mixture components, and temporal dilution or variation in concentration of mixture components in the exposure medium (water, diet) or in tissues of test organisms;
- Do not assume that correlation proves causation - it does not;
- Consider all data and do not ignore inconsistencies that may well provide critical information to assist in determining causation;
- Determine causation relative not just to the mixture but to its constituent components, including biotransformation products - such products can sometimes be more toxic than the parent compounds;
- Conduct confirmation studies, such as bio-effect directed fractionation, or tests with individual or surrogate mixtures of mixture components, and/or perform modeling based on known mechanisms, such as molar or TU addition; and
- Discuss any and all uncertainties explicitly and transparently - do not overstate the cause of the toxicity when only inference has been established by focusing on a portion and/or portions of a mixture.

4.3.5 Realism in toxicity tests

The concern expressed in this review for maintaining constant concentrations of oil during toxicity tests runs counter to concerns that tests reflect the 'reality' of oil spills, when concentrations of oil vary enormously, both spatially and temporally . The 'spiked' C ROSERF protocol for preparing and testing solutions of WAF and CEWAF models the rapid loss of hydrocarbons from the water column at actual spills by standardizing one decay rate of oil concentrations in test solutions (Singer et al. 2000). The method was described as more realistic than tests in which concentrations remain relatively constant. However, given the site - and spill -specific fate and behaviour of oil, standardizing tests to one time - varying scenario creates the same lack of realism as a constan t exposure scenario. Interpreting exposure concentrations that vary temporally is difficult and presents significant problems for predictive models of toxicity used in risk assessments that assume constant exposure concentrations (Section s 4.1.4; 4.3.1). Predictive models treat toxicity as a fixed reference point that can be modified by known environmental conditions to estimate effects under natural conditions.

Given that results of toxicity tests are strongly influenced by protocols chosen for preparing a nd testing oil solutions (Redman and Parkerton 2015), laboratory tests can never be realistic. However, methods can be standardized to provide data comparable among labs, species and oils. Thus, NRC (2005) suggested that the CROSERF method be refined to:

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- Standardize mixing energies for WAF and CEWAF;
- Prepare gradients of test concentrations by diluting stock solutions instead of preparing unique stock solutions for each concentration;
- Prolong the exposure to measurable concentrations of hydrocarbons;
- Replace nominal exposure concentrations with measured concentrations of TPH s and/or TPAHs, and report the time during the exposure when concentrations are measured;
- Include treatments to test the potential for photo-enhanced toxicity; and
- Substitute open test chambers for sealed test chambers to allow volatiles to evaporate during the test.

Many of these recommendations have been followed in recent studies. *For ERAs, effects under realistic conditions could be estimated by applying toxicity data to the unique site-specific circumstances of each potential or actual oil spill.*

Recommendation: Research is needed to determine the relationship between toxicity and exposure time, and the nature of delayed toxicity following acute exposures (Section 4.3.2).

4.4 Research Needs and Recommendations

The following is a synthesis of research needs identified in previous sections. Many involve the development and standardization of protocols for lab toxicity testing, post -spill field assessme nts and monitoring of impacts.

4.4.1 Research Needs

Research is needed:

- At 'spills of opportunity' and experimental oil spills to identify effects on aquatic species at the level of populations, communities and ecosystems due to the acute and long-term toxicity of spilled oil;
- On how the behaviour and fate of fresh and weathered oil in different ecosystems interacts with the habitat selection and unique life history traits of aquatic species to control species -specific oil exposure and toxicity. These interactions are particularly important for dilu ted bitumen (dilbit) relative to conventional oils, and northern and Arctic ecosystems relative to temperate ecosystems;
- To understand how different oil types (e.g., dilbit) and spill-control agents (e.g., chemical dispersants) affect aquatic species under different environmental conditions, including interactions with contaminants from municipal, industrial and agricultural effluents;
- On mechanically- and chemically-dispersed oil to determine how oil droplets affect the exposure and toxicity of oil to different species and life stages of aquatic organisms. These studies include a need to develop analytical methods to reliably discriminate droplet oil from dissolved oil. Studies on the distribution and effects of dispersed oil under exposure conditions simi lar to those that may be encountered during response operations (including subsurface injection of dispersants) are of particular importance;
- On strategies and protocols to recover spilled oil that avoid or minimize habitat damage, methods to restore dama ged aquatic and riparian habitats and models to define the balance between the environmental costs of natural remediation and oil spill cleanup; and
- On methods for finding and measuring spilled oil in different compartments of freshwater and marine ecosystems, for measuring exposure and toxicity of oil to aquatic species *in situ*, and for assessing the structure and function of ecosystems that reflect the extent and time to recovery following oil spills;
- On the contamination of ground water due to soil cont amination and the entrainment of oil droplets into bed sediments of rivers by surface water-ground water exchanges;

- To develop standardized oil toxicity test methods for all classes of aquatic organisms to enable reliable comparisons of toxicity among oils , species and different environmental conditions (e.g., salinity, temperature), including rapid changes in oil concentrations in water; and
- To refine models that predict acute and chronic toxicity to aquatic species, particularly to assess the uncertainty caused by violating the assumptions underlying these models and to reduce the variance of model predictions.

4.4.2 Operational/preparedness Needs

Baseline monitoring is needed in areas where oil is shipped or produced to resolve uncertainties about 'cause and effect' should a spill occur. Baseline monitoring should include:

- The physical and chemical characteristics of aquatic ecosystems; and
- The ecology, population dynamics, productivity, population demographics , abundance and seasonal and spatial use of habitat by locally-resident and transient aquatic species.

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CHAPTER 5: MODELING OIL SPILLS IN WATER

Abstract

Oil contains thousands of compounds and its properties can change drastically as natural weathering processes proceed immediately following a spill. For this reason, earlier modeling of oil behaviour and transport was phenomenological with a heavy reliance on experimental observations. Since the early 1980s, advances in oil spill modeling have been largely focused on oil dispersion (formation of oil droplets), formation of oil particle aggregates (especially after the *Exxon Valdez* oil spill in 1989), emulsification, evaporation and general transport in open waters, as well as other types of ecosystems. As a consequence of the Deepwater Horizon spill and emerging concerns over the environmental risks associated with offshore oil and gas activities in frontier regions, more advanced numerical models have recently emerged to better predict various processes, especially in situations where no direct measurements could be made, such as in deep water or in the Arctic. Major modeling advances have emerged in the areas of dispersion, biodegradation, dissolution and oil-in-ice behaviour. To improve our understanding on the influence of environmental factors on the fate and behaviour of spilled oil, modeling efforts continue to be needed in the area of oil-in-ice, the prediction of oil dispersion by waves, oil droplet formation from blowouts, formation of oil particle aggregates and the biodegradation of oil droplets under various environmental conditions, including temperature, salinity, nutrients and light (to elicit the significance of photooxidation processes).

Introduction

In modeling, it is often said that a large spill of oil is actually multiple spills. This is because the oil properties change with time and distance from the source and the tools adopted near the source could be ineffective 20 km from the source. Also the tools used on day one could become obsolete on day 10 because of oil weathering or transport to a different type of environment (e.g., from open water to beaches). In addition, oil spills could cover large areas and it is cost prohibitive to model all processes occurring at the smallest scales. Yet, the impact is localized, dependent on the concentration at a particular receptor. Earlier modeling efforts of oil spills relied on using simple formulations for oil weathering that are calibrated to experimental studies. The resulting formulations have been used in subsequent oil spill models. However, no rigorous field validations were available or forthcoming due to lack of experimental replication or accurate characterization of the environmental conditions (such as wave energy, etc.). Improvements on these formulations have taken place in recent years in various categories, including emulsion, dispersion and biodegradation of oil. However, more work is needed, including the formation of oil particle aggregates (OPAs), a fraction that was previously not accounted for in mass balance calculations, which also influences the transport (e.g., sedimentation rates), fate (e.g., oil biodegradation rates) and effects (e.g., bioavailability) of the residual oil.

The change of oil properties with time is known as 'weathering' and it affects the subsequent transport and behaviour of oil. Chapter 2 discusses these effects in detail. For this reason, this chapter addresses first the modeling of the fate of oil due to weathering and then the modeling of the transport of oil in various compartments of the environment: lakes, rivers, shorelines, wetlands, etc. To facilitate understanding of the information in this chapter, most modeling equations discussed were not included in the write-up; rather, the equations were explained by discussing the terms in simple English.

5.1 Modeling of Oil Weathering

When oil is spilled into the environment, its physical and chemical properties change over time in a process known as weathering, which includes the following processes: spreading, evaporation, dissolution, dispersion, photooxidation, emulsification, OPAs and biodegradation (**Figure 5.1**). All of these are discussed in detail in Chapters 2 and 3. Analyzing and quantifying the degree of oil weathering

(i.e., oil fate) is a necessary step to determining oil transport and per sistence in the environment. For this reason, this chapter starts by addressing oil weathering.

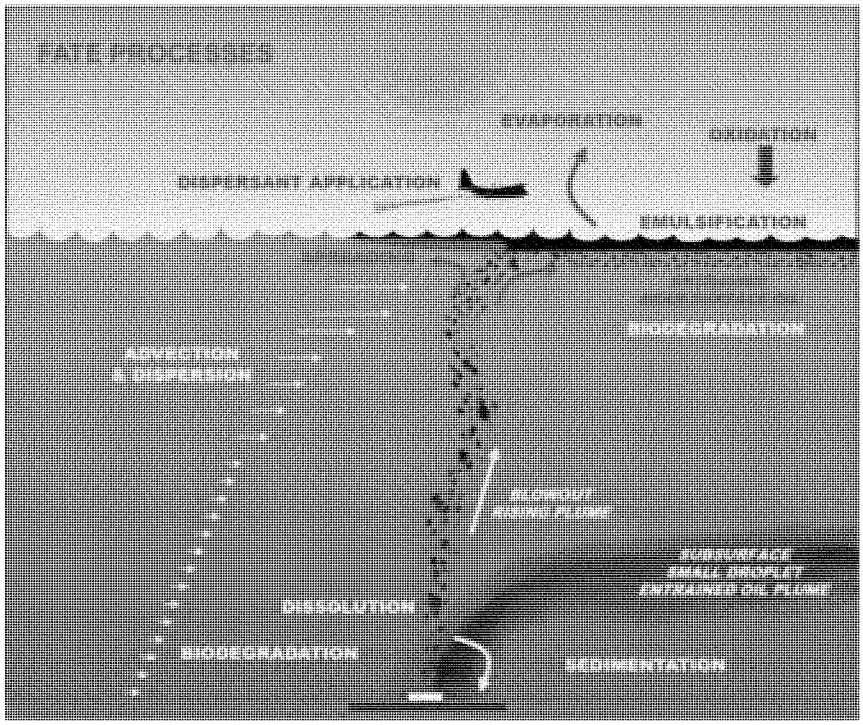

Figure 5.1 Illustration of the fate process of oil spills in water *. Image from Water Resources and Modeling Group of ERM Inc. (http://gemss.com/cosim.html).*

5.1.1 Oil evaporation

The modeling of oil evaporation is based on equations originally developed for water evaporation (Sutton 1934, Brustaert 1982) , where the exchange of water between the waterbody and the atmosphere was modeled using a mass transfer formulation in which the exchange depends on the water vapour pressure in the atmosphere and on a mass -transfer coeff icient that represents the hydrodynamics at the water - atmosphere interface (Brustaert 1982) . Sutton (1934) developed a mathematical expression for water evaporation having the terms mass transfer coefficient, concentration of the evaporating liquid, wind speed, size of the water body and a ratio of kinematic viscosity (defined in Chap ter 6) of the air to the molecular diffusivity in air. Sutton's formula shows that as the wind speed increases, water evaporation increases, which confirmed empirical observations.

Most of the modeling of oil evaporation relies on using the vapour pressure of various oil components based on distillation data and average boiling points. Blokker (1964) presented one of the earlier models under this category. Yang and Wang (1977) developed a model that accounted for the role of the slick area in affecting evaporation. However, one of the most widely used models for oil evaporation was developed by Stiver and Mackay (1984) . That model included terms for the volume fraction evaporated, absolute temperature, a ratio called the 'evaporative exposure', and constants obtained based on experimental data. Similar models include those developed by Tkalin (1986) and Hamoda et al. (1989) . The latter correlated the mass transfer coefficient to the API gravity of oil and the salinity of the water and found that oil evaporation increased slowly proportional to salinity. Bobra (1992) compared the Stiver and Mackay (1984) model to extensive experimental data of oil evaporation and reported that the model performed well for the first eight hours, but overestimated long-term evaporation in most cases.

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In a series of later studies, Fingas (2011; 2013) challenged basing oil evaporation models on water evaporation, arguing that the evaporation of oil is not limited by the air boundary layer, but by the diffusion of oil components to the interface between the oil and the atmosphere. The author also noted that the vapour pressure of water is much lower than most of the hydrocarbon compounds of interest for evaporation. Thus, the author concluded that the limiting factor for oil evaporation was in the liquid oil and not in the air above it. The studies included evaporation experiments of diesel, ASMB (Alberta Sweet Mixed Blend) oil and water in wind tunnels at different wind speeds and observed that the wind speed increased the evaporation of water. However, the increase of oil evaporation with wind speed occurred only when comparing to the no-wind condition. All non-zero wind speeds gave essentially the same oil evaporation rate. Based on these results, Fingas (2013) developed two different empirical equations that predict evaporation of oil based on the properties of the oil, one for diesel and one for heavier oils. Both equations indicate that evaporation decreases with time under given environmental conditions.

A study by King et al. (2013) in a flume tank exposed to the atmosphere found that the density of the Access Western Blend increased to more than that of fresh water while the density of the Cold Lake Blend remained lower than that of fresh water. They used a hyperbolic law for estimating the increase in mass density with time of the two products.

5.1.2 Oil dissolution

In general, dissolution is not a major mechanism for oil spilled on to the water surface, as most of the components that dissolve have high vapour pressures and thus evaporate quickly. Thus, evaporation and dissolution for surface oil are competitive processes with evaporation the dominant one (French-McCay 2004). However, dissolution could become important for oil away from the atmosphere, such as from underwater releases, namely the Deepwater Horizon (DWH) spill (Camilli et al. 2012) or in the subsurface, such as in the aquifer of Bemidji oil spill site (Essaid et al. 1995; Cozzarelli et al. 2010; Ng et al. 2014). The compounds that dissolve rapidly and result in measurable concentrations in the water column are the monoaromatics, also known as BTEX (benzene, toluene, ethylbenzene and xylenes). Naphthalene, a PAH, also has a high solubility in water, and so are short chain alkanes, such as methane, ethane, propane and butane (Ryerson et al. 2012).

> Oil evaporation dominants the mass transfer rates for surface oils, while for underwater releases and in the aquifer, oil dissolution becomes important. For underwater releases, droplet size distribution is the most important parameter affecting the oil dissolution rate.

Mackay and Leinonen (1977) and Mackay et al. (1980) conducted theoretical investigations of mass transfer at the oil-water interface and concluded that the dominant mechanism of transfer of soluble hydrocarbons into the water column during an oil spill is by dispersed oil droplets rather than surface slicks. Thus, droplet size distribution is the dominant factor in determination of the oil dissolution rate in subsurface oil release. Small droplets (e.g. ~ 100 μm in diameter) ascend slowly through water and could remain in the water column for days or months. This buoyancy contributes significantly to oil dissolution. Large droplets (e.g., greater than a few hundred μm in diameter) rise to the water surface rapidly, so dissolution from those droplets is relatively inconsequential (French-McCay 2004). Mackay et al. (1980) showed that dissolution from surface oil is negligible.

The dissolution of oil differs from that of a pure compound due to the competition of high solubility compounds reaching the oil-water interface and subsequently the water column. In addition, polar compounds in the oil, such as the resins and asphaltenes, tend to migrate to the oil-water interface. However, they tend to remain at the interface due to their low solubility in water. Therefore, these compounds tend to lower the dissolution rate of the remaining oil components in the droplet into the water column.

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5.1.3 Photooxidation

Photooxidation results from exposure of spilled oil to sunlight, which causes some components in the oil to oxidize, potentially facilitating their chemical and biological degradation. Yang et al. (2015) studied the photochemical behaviour of crude oil in a simulated freshwater environment and found that lower molecular weight alkanes were photooxidized more rapidly than the higher molecular weight compounds. They also found that PAHs can be photooxidized more readily than alkanes, which is consistent with prior studies (Garrett et al. 1998; Maki et al. 2001; Prince et al. 2003; Radović et al. 2014; D'auria et al. 2009). Prince et al. (2003) also report that the majority of aromatic fractions were converted to resins and asphaltenes. However, a more recent study (Aeppli et al. 2012) indicated that photooxidation produces oxygenated hydrocarbons that might persist in the environment and can produce carbonyl compounds (compounds with a C=O group such as carboxylic acids, aldehydes, amides, etc.) and alcohols, which are soluble in water and thus can be carried by water flow (Chapelle 2001).

5.1.4 Oil emulsification

When oil persists on the water surface for hours and days, it tends to incorporate water droplets within it, forming water-in-oil emulsions. This process is the opposite of oil dispersion where oil droplets are incorporated into the water column. The water-in-oil mixtures are grouped into four classes: stable, meso-stable, unstable and entrained water (Fingas and Fieldhouse 2004). The characterization is based on the persistence of the water-in-oil mixture and also visual observation. Stable emulsions, called mousse (Chapter 3), tend to be semi-solid material with a reddish-brown (even orange) color and can persist for weeks. Meso-stable emulsions tend to be liquid and brownish to black in colour and tend to persist for several days to a week under laboratory conditions. However, they can persist much longer in the environment. The remaining two classes are black in colour, and the water content of these water-in-oil emulsions varies. Stable emulsions contain around 80% water. Meso-stable emulsions initially contain around 70% water, and the amount of water decreases to around 30% after a week. The unstable and entrained-water classes contain usually around 10% water by volume (Fingas and Fieldhouse 2004), and are common in low viscosity oils (viscosity less than 20 cp [1]) or high viscosity oils (viscosity higher than 1,000 cp, such as weathered diluted bitumen).

Emulsification appears to increase as the resins and asphaltenes in the oil increase. These latter compounds act as surfactants and thus situate at the oil-water interface (Bobra 1992). Both asphaltenes and resins comprise polar compounds. Observations (Fingas et al. 1996, 2000, 2002) revealed that the weight fraction of asphaltenes was typically higher than 7% in a stable emulsion, and that the weight fraction of asphaltenes and resins was higher than 3% in a meso-stable emulsion. The presence of waxes (high molecular weight alkanes) was found to also promote emulsification but to a lesser extent than asphaltenes and resins. Thus, the stability of water-in-oil emulsions seems to depend primarily on the asphaltene content. However, it is thought that resin molecules, which are smaller than asphaltene molecules, tend to initially locate at the oil-water interface, causing the formation of the emulsion, but that asphaltenes subsequently migrate toward the interface and render the emulsion stable. In that sense, asphaltenes seem to be a stronger surfactant than the resins. Also, in fairly general terms, emulsification is promoted by the viscosity of the oil (i.e., oils that have moderate to high viscosity tend to emulsify more than lighter oils). Thus, emulsion formation increases with weathering of the oil, especially due to evaporation and dissolution. Both processes cause the viscosity of the oil to increase. Therefore, emulsification models depend on the outcome of the evaporation and dissolution models.

Traditional modeling of emulsification relied on predicting three parameters, the water content in the emulsion, the class of emulsion (discussed above) and the resulting viscosity of the emulsion (Fingas and Fieldhouse 2004; Xie et al. 2007). The water content is important because in a stable emulsion, it is 80%

[1] *Centipoise (cp) is a measurement unit of dynamic viscosity in the cgs (cm-g-sec) system of units.*

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by weight, which means that an emulsion contains approximately five times the initial volume of the oil. This poses major challenges for removing and treating the emulsion in the environment. Mackay et al. (1980) developed an equation to estimate the water content of a water-in-oil emulsion containing two rate constants for water incorporation and a third rate constant that depends on the coalescing tendency of the emulsion. Mackay and Zagorski (1982) suggested a stability index, which is calculated from the asphaltene and wax content in the oil, other oil components and temperature. Thus, for a content of asphaltene > 7% the stability index is > 1.22, and for the content of asphaltenes and resins > 3% the stability index declines to 0.67. Values of the stability index between 0.67 and 1.22 represent a meso-stable emulsion, while values less than 0.67 represent an unstable emulsion. Zeidan et al. (1997) proposed a chemical composition index to determine the class of emulsion. The model contains terms for weight fractions of asphaltenes, resins and waxes. The dynamic viscosity of the emulsion can be estimated by the method of Mooney (1951). Xie et al. (2007) also considered the demulsification of the emulsion and represented it using an exponential decay curve for meso-stable and unstable emulsions. They noted that a minimum amount of energy was commonly needed to initiate the emulsion, and they also developed estimation methods based on the sea state.

In a departure from the literature on emulsification modeling, Fingas and Fieldhouse (2004) argued that the constituent oil fractions are not sufficient to fully predict the emulsification class. For example, diluted bitumen contains more than 20% by weight of either asphaltenes or resins, yet it does not form an emulsion because of its high dynamic viscosity (more than 10,000 cp). They considered an extensive database of emulsions and developed empirical relations to predict the classes of emulsions, and the corresponding water content and viscosity.

Emulsification was found to reduce oil evaporation and dissolution. Experiments by Ross and Buist (1995) revealed that the intensity degree of oil evaporation decreases with increasing water content and slick thickness, both due to emulsification. Xie et al. (2007) investigated these findings and developed an empirical model for oil evaporation from an emulsion. The model contained terms for the weight fraction of water, the oil evaporation from pure oil and the evaporation from the emulsion.

5.1.5 Oil dispersion

For surface oil spills, wave action breaks the oil slick into droplets (i.e., oil dispersion) propelling them into the water column. An empirical formula was developed by Delvigne and Sweeney (1988) and Delvigne (1994) for dispersion. The important terms in the formula were the particle size, the particle size interval, the entrainment rate per unit surface area of oil particles with particle sizes in a specific interval around the particle size term, an empirical constant dependent on the oil type and weathering state, the energy dissipated during breaking waves per unit surface, the fraction of surface are a covered by oil, and the fraction of sea surface hit by breaking waves per unit time. The energy dissipated term and the sea surface fraction of breaking waves per unit time were further modeled using two additional semi-empirical formulas containing terms for root-mean of wave heights, wind speed and threshold wind speed for generation of breaking waves.

In another model, Mackay and Leinonen (1977) assumed that the dispersion of oil from the slick was the sum of two rates: a breaking wave rate and a non-breaking wave rate. This model included terms for the volume fractions of oil droplets permanently dispersed by non-breaking and breaking waves, the slick thickness and the fraction per second of slick area forced into the water column by the breaking waves.

Probably the most appealing aspect of the Delvigne and Sweeney (1988) model is that it estimates the amount of oil entrained based on the wave energy. Another appealing aspect of this model is the ease-of-use and low computational cost. However, no validation of the model has been made in field situations due to the large scale of natural systems and variability in the sea.

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Zhao et al. (2014 a,b) developed the model VDROP, a comprehensive model that estimates the droplet size distribution (DSD) based on the properties of the oil and the hydrodynamics of the system. While most models of its type consider the interfacial tension to be the only force resisting breakup, the VDROP model allows for situations where the viscosity of the oil resists the breakup of the droplets, which occurs when the oil is highly viscous and/or when dispersants are used, causing the interfacial tension to decline. Zhao et al. (2014a) considered that, after each breaking wave event (assumed every two minutes), droplets smaller than 100 µm were driven into the water column away from the surface. The energy dissipation rate in the model can be roughly estimated based on the sea state (i.e., amount of breaking waves).

5.1.6 Oil particle aggregation

As oil approaches coastal regions it tends to interact with the higher load of suspended particles in the water column to form OPAs, which move differently from oil droplets or particles alone (**Figure 5.2**). The suspended particles include mixtures of silt, clay, detrital organic matter and algae or plankton, whose concentrations near shorelines are much higher than encountered by oil offshore, especially if the oil came from a surface spill. There are two major types of OPA: 1) oil droplets coated by small particles resulting in the so-called Pickering emulsion (Le Floch et al. 2002; Frelichowska et al. 2010); and 2) large particles adsorbing oil onto them and trapping oil between them (Stoffyn-Egli and Lee 2003). The first type is the most common and results in a larger volume of trapped oil, and sometimes can consist of many oil droplets binding together forming a large OPA (Le Floch et al. 2002; Omotoso et al. 2002). In this type, the oil droplets of interest are typically larger than 50 µm, whereas the particles are typically smaller than a few µm.

> Alternative terms to OPA have been used, such as OMA (oil mineral aggregates) and OSA (oil suspended particulate matter aggregates). The term OPA was coined by the Boufadel group while addressing the diluted bitumen spill in the Kalamazoo River in Michigan 2010. The group argues that existing terms are restrictive as the particulates do not have to be minerals, and the suspension state is not an intrinsic property, rather depends on the hydrodynamics.

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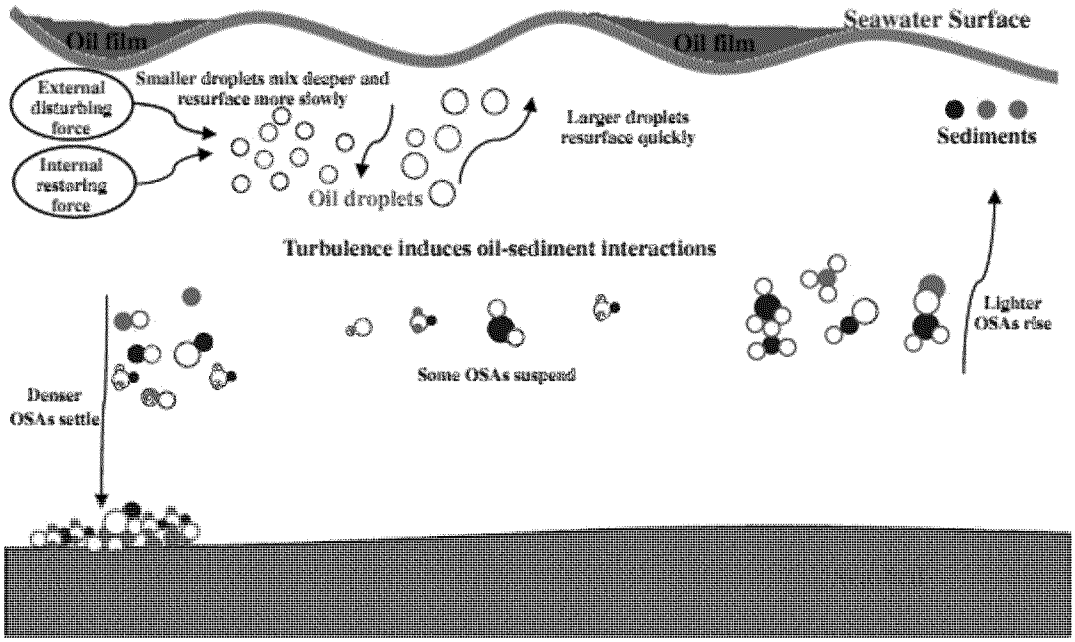

Figure 5.2 Formation and movement of various types of OPAs in marine systems. *The term OSA implies (oil suspended aggregates). The term OPA comprises OSA and OMA (as developed by the Kenneth Lee group). This image is reprinted from Gong, Y., Zhao, X., Cai, Z., O'Reilly, S.E., Hao, X. and D. Zhao. 2014. A review of oil, dispersed oil and sediment interactions in the aquatic environment: Influence on the fate, transport and remediation of oil spills. Marine Pollution Bulletin 79: 16-33. Copyright Elsevier (2014).*

Of interest in modeling the OPAs is the critical particle concentration, which is the concentration that results in complete coverage of the oil droplet, observed to be around 30 to 50% of the oil concentration expressed in the same units (Ajijolaiya et al. 2006). For example, if the oil concentration in the water column is approximately 100 mg/L, then a sediment concentration of 30 to 50 mg/L would ensure complete coverage of the oil droplets.

The formation of aggregates depends on the viscosity and adhesive properties of the oil droplets, the surface area and chemical and physical properties of the particles, and the salinity of the water and other factors such as the presence of dispersants (Lee 2002; Khelifa et al. 2005). High viscosity oils (dynamic viscosity >500 cp) or those with high adhesion (>50 g/m ^{2}) (Jokuty et al. 1995) tend to bind to sediments rapidly. The efficiency of trapping increases with the hydrophobicity of sediments until it reaches a maximum and then decreases (Sun et al. 2010). This is because as the particles become more hydrophobic, they tend to attach more closely to oil. However, if they become too hydrophobic, they tend to clump together and thus their chance of adhering to oil decreases. In addition, the increase in salinity enhances OPA formation, and enhancement is high when the salinity increases to >1,000 mg/kg (Sun et al. 2010).

Common modeling approaches for the OPAs rely on estimating the trapping efficiency by empirical expression fitted to experimental data. This is unlike the approach that has been used for modeling oil dispersion in chemical engineering (Tsouris and Tavlarides 1994) or in oil spill mitigation (Zhao et al. 2014). A justification for the simplification in modeling OPAs is probably due to the fact that, unlike oil droplet models where only oil droplets interact with each other, an OPA model would require the interaction of droplets, particles and OPAs, resulting in an excessive number of equations to capture all interactions. Nevertheless, existing OPA models cannot predict based on the fundamental properties of the oil or particles, and they constitute a challenge when predicting the behaviour of oil in systems with

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high sediment (or particle) content, such as streams (Fitzpatrick et al. 2015) and shorelines (Owens and Lee 2003).

5.1.7 Oil biodegradation in open water

Since the biodegradation of oil depends on the oil-water interfacial area (Atlas and Hazen 2011), the biodegradation of slicks is considered negligible in comparison to that of oil droplets. Therefore, for oil spills on the water surface, dispersion is a key factor affecting biodegradation. The approach adopted in NOAA's *Automated Data Inquiry for Oil Spills* (ADIOS2) model consists of treating the oil as various classes (C_4-C_{12} alkanes; 2-3 ring aromatics) to allow for the reduction of the size of oil droplets following biodegradation (Viveros et al. 2015). This is a simple approach as the ADIOS2 model is intended for short-term prediction, mainly during the response to an oil spill. A recent work by Yassine et al. (2013) modeled oil biodegradation using Monod kinetics and a quasi-steady state approximation for the dissolution of low solubility hydrocarbons in the water column. They reported good agreement with data. Vilcáez et al. (2013) assumed that the microorganisms covered oil droplets completely. The simulation results revealed that small oil droplets biodegraded faster due to their larger surface area per unit mass. Their model suggested that oil droplets biodegraded faster than dissolved oil components due to their assumption of complete microbial coverage of oil droplets. However, they provided no data to confirm their assumption.

5.1.8 Oil biodegradation within beaches

When oil reaches the shoreline, it percolates into the subsurface when the tide falls, and after a few days a residual amount of oil remains entrapped within the sediments by capillary forces, regardless of the action of the subsequent tides and waves.

Numerical models have been used to predict biodegradation of petroleum hydrocarbons since 1980. However, most of them aimed at modeling the biodegradation of dissolved hydrocarbons (Borden and Bedient 1986; Essaid et al. 1995; Clement 1997). In the model of Borden and Bedient (1986), the biodegradation of dissolved hydrocarbons was assumed to be limited by the availability of oxygen. One- and two- dimensional models were used to investigate the effect of microbial kinetics and oxygen transfer on hydrocarbon biodegradation. Essaid et al. (2003) used the U.S Geological Survey (USGS) solute transport and biodegradation code BIOMOC coupled with the USGS universal inverse modeling code UCODE to quantify the BTEX dissolution and biodegradation at an aquifer site located in Bemidji, MN. Essaid et al. (1995) used a two-dimensional model to simulate the sequential aerobic and anaerobic biodegradation of volatile and non-volatile fractions of dissolved organic carbon. Clement (1997) developed a modular computer code called RT3D to describe reactive multispecies transport in three-dimensional groundwater systems with consideration of multiple electron acceptors for hydrocarbon biodegradation. These studies also accounted for the hydrocarbons in an immobile oil phase by adding oil sorption under equilibrium (Borden and Bedient 1986; Essaid et al. 1995) or non-equilibrium conditions (Clement 1997) into their models. The model SEAM3D (Widdowson et al. 2002) was developed to account for sequential electron acceptors in aquifers contaminated with dissolved hydrocarbons. In this model, the hydrocarbons used molecular oxygen as an electron acceptor, followed by nitrate, manganese, iron, sulfate (sulfate reducing), and finally carbon dioxide (methanogenic). SEAM3D was used to evaluate the potential biodegradation of oil trapped in the sediments of a beach in Grand Isle, LA (OSAT 2011), and it was found that the dissolution of oil was a limiting factor for the migration of oil off site. However, all of the aforementioned models applied to dissolved hydrocarbon or relied on physical dissolution from the immobile/oil phase into the mobile phase for biodegradation to occur. Thus, the biodegradation of the low solubility hydrocarbons, which occurs by microorganisms colonizing the oil-water interface, cannot be effectively accounted for.

Nicol et al. (1994) presented a frame work for one -dimensional modeling of biodegradation of residual low solubility petroleum hydrocarbons in fully saturated porous media. The biodegradation of residual oil was limited by electron acceptor (dissolved oxygen) and nutrients (nitrogen and phosphorus), which were transported through the porous medium by advection and dispersion. El-Kadi (2001) developed a model for hydr ocarbon biodegradation in tidal aquifers considering the impact of the water content and temperature on microbial activities. Both dissolved and trapped hydrocarbons were considered. The model assumed that trapped hydrocarbons were immobile in the porous m edium and were subject to the same biodegradation mechanisms as dissolved hydrocarbons. For modeling trapped/residual hydrocarbon biodegradation, both studies (Nicol et al. 1994; El-Kadi 2001) assumed that the hydrocarbons dissolved in the aqueous phase by biosurfactants produced by the oil degrading bacteria are directly metabolized by the bacteria.

The model BIOB (Geng et al. 2013,2014 ; Torlapati and Boufadel 2014) was developed to simulate the biodegradation of low solubility hydrocarbons in porous media. In the model, the rate of hydrocarbon biodegradation was considered proportional to biomass growth (Rittmann and McCarty 2001 ; Tchobanoglous et al. 2002). In order to consider the washout of hydrocarbons in a beach environment, a hopane-normalized term was added to the equation of hydrocarbon removal followi ng the work of Venosa et al. (1996) . The biomass growth rate is assumed to be limited by the availability of hydrocarbons and nitrogen, expressed as a dual -Monod formulation (e.g., Molz et al. 19 86). In field environments, the size of the microbial population in a porous medium is constrained by factors , such as pore space, production of inhibitory metabolites and sloughing of microbial biomass (Weise and Rheinheimer 1977; Kazunga and Aitken 2000; Rønn et al. 2002). To avoid unreasonably high microbial populations, Geng et al. (2014) used the formulation of Kindred and Celia (1989) and Schirmer et al. (2000) introduced a term to the model to account for the decrease in biomass accumulation when th e microbial concentration approaches its maximum value. The BIOB model was calibrated to laboratory column data reported in Boufadel et al. (1999) and Venosa et al. (2010) and to field data reported in Venosa et al. (1996). The model was able to reproduce the experimental data of oil, oxygen consumption, carbon dioxide production and biomass as represented by the Most Probable Number. The BIOB model was also capable of capturing the impact of nutrient concentration of microbial growth and hydrocarbon biodegradation.

5.2 Oil Transport

In this section, the transport of oil is addressed in systems of increasing complexity. First the spreading of oil on the water surface is considered. This is followed by a discussion of the transport of oil on lakes, where wind and waves are the major conveyance mechanisms. Then, the transport of oil in streams and rivers is examined, which adds the complexity of the interaction of the main channel with stagnant zones and with the hyporheic zone (the subsurface surrounding the strea m). Finally, the transport of oil in estuaries is reviewed where vertical salinity gradients could confine the dispersed and dissolved oil to the upper portion of an estuary.

5.2.1 Spreading

Fay (1971) and Hoult (1972) developed physical arguments for the spreading of oil spilled on a calm sea. They explained that initially the oil spreads due to gravity because oil rests higher than the water surface, and afterwards it spreads due to the interfacial tension between oil and water. The water inertia resists the spreading of the oil in the initial phase of the spill, and the water viscosity becomes the main r esisting force after the initial phase. However, when wind and/or waves are present, the modeling of surface spreading would need to account for the elongation of the oil along the wind direction (Lehr et al. 1984). Thus, instead of a circular spill based on the Fay's analysis, one would obtain an ellipse with the long axis aligned with the wind direction.

5.2.2 *Transport in lakes and sea*

The movement of oil on lakes is driven to a large extent by the action of winds because the current velocity is small. Wind affects the movement of oil on water in three ways:

- Wind can transport oil floating on the water surface without affecting the water beneath it. The speed of oil movement carried by the wind could be up to 6% of the wind speed and is typically assumed to be 3 to 4% in some models (ASCE 1996; Barker 2011; Boufadel et al. 2014);
- Wind over a fetch of say 5 km can generate waves, which can alter the trajectories of oil. As the wave propagates, the water velocity beneath it changes: the water velocity is forward (in the direction of wave propagation) under the wave crest and backward under the trough . Thus, floating objects, such as an oil slick on the water surface or oil droplets dispersed in the water column, move forward when the crest passes and they move backward when the trough passes. However, the forward movement is greater than the backward movement resulting in a net forward transport known as the Stokes drift (Dean and Dalrymple 1984) . The maximum value of the Stokes drift is at the water surface, and in general it decreases exponentially with depth. Waves also break up the oil slick into oil droplets, causing dispersion, and propels them into the water column. The subsequent transport of oil droplets would depend on the size of oil droplets, as noted by Elliot et al. (1986) and later by Boufadel et al. (2006, 2007) . These studies explained the comet shape of oil spills, wherein a large mass of oil propagates ahead of a tail. The large oil droplets (which typically hold most of the oil mass) have a higher buoyancy and thus remain near the water surface, and get advected further forward by the Stokes drift than smaller oil droplets that tend to be deeper due to diffusion and wave motion; and
- Wind can generate Langmuir cells (Langmuir 1938 ; Leibovich and Ulrich 1972) , which are circulation cells within the water column perpendicular to the wind propagation direction. They would transport the oil along windrows and cause it to down -well into the water column. A salient example of this was the 1993 *Braer* oil spill off the coast of Shetland, Scotland, where the oil disappeared off the water surface to reappear hours later, and the movement w as attributed to Langmuir cells (Farmer and Li 1994).

5.2.3 *Transport in streams and rivers*

The transport of the bulk of spilled oil in a stream or river can be easily predicted based on the stream discharge and stream morphology, which provide the hydrodynamic properties needed by the model, such as stream velocity and the turbulent diffusion co efficients (Fischer et al. 1979 ; Rutherford 1994) . However, a small but non -negligible amount of oil remains entrapped in dead or stagnant zones near the banks of the river and in the hyporheic zone, which is the subsurface surrounding the stream (Jones and Mulholland 1999). These zones behave as transient storage zones retaining wate r and oil from the main channel and releasing them at a slow rate, thus extending the residence time of an oil spill in the stream. The transport of oil in rivers could also be affected by the regional groundwater table, and whether the reach is gaining or losing water (Jones and Mulholland 1999). Field studies revealed that groundwater in rivers gaining groundwater could conf ine plumes into the main channel, which would increase their downstream transport due to the larger velocities near the thalweg[2] of streams (Ryan and Boufadel 2006).

Hyporheic flows (i.e., the flow between the main channel and the hyporheic zone [3]) are very important ecologically, as the hyporheic zone provides a sp awning ground for fish. Fish embryos may be adversely affected by hydrocarbons, as hyporheic outflows from oil -contaminated rivers may carry dissolved and particulate oil into the hyporheic zone. Hyporheic flows also carry oxygen into sediments and remove metabolic waste products, such as ammonia, due to the metabolism of communities of bacteria, fungi and

[2] *The thalweg is a line along the stream length connecting the lowest points of the streambed.*
[3] *The hyporheic zone is a region beneath and alongside a stream bed, where mixing occurs between the shallow groundwater and surface water.*

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invertebrates that inhabit the hyporheic zone. If down-welling in shallow water transports oil into the hyporheic zone, particulates may be trapped as free-phase oil in the interstitial spaces between sediment particles and act as a source of dissolved hydrocarbons from oil-water partitioning (Hodson et al. 2011). In a field study of a tidal stream abutting Prince William Sound, Carls et al. (2003) reported hyporheic flows that would carry dissolved hydrocarbons to pink salmon (*Oncorhynchus gorbuscha*) spawning in the stream.

Hyporheic flows are generated by two non-exclusive mechanisms: 1) the irregular bed form generates a pressure gradient along the hyporheic zone, causing stream water to enter the subsurface upstream of a dune and to exit it at the lee side (Elliott and Brooks 1997; Tonina and Buffington 2007, 2009; Buffington and Tonina 2009); and 2) the subsurface heterogeneity in a streambed (Ryan and Boufadel 2006, 2007a,b) can engender hyporheic flows even though the bathymetry is flat (Ryan and Boufadel 2007a,b). Hyporheic flows are also affected by the groundwater table (Harvey and Fuller 1998) as they increase under baseflow conditions (i.e., low water table) and decrease under high flood conditions (i.e., high water table). Thus, an oil spill during baseflow would introduce oil more into the subsurface than an oil spill during high flows.

In many situations, a better understanding of sediment transport in rivers is needed to understand the transport of oil and OPAs. Sediment transport in rivers occurs through three complementary mechanisms:

- Wash load consists of very fine particles that are not present in abundant concentrations on the bed. Examples include suspended organic matter. Therefore, knowledge of bed material composition does not allow one to predict wash load transport very well;
- Suspended load is the load that is suspended in the water column but still interacts with the bed. The interaction usually occurs at ripples causing the shear stress to increase and sediments to be suspended from the lee side of the ripples; and
- Bed load has continuous contact with the bed and has a direct relation to the bottom stress in rivers. Both suspended and bed loads are commonly predicted using sediment transport models. Thus, to predict the formation of OPAs, sediment transport models might not provide a complete description of the transport, as they focus on the suspended load and bed load, while the formation of OPAs depends also on the wash load.

The types of OPAs greatly affect their transport. It is worth noting that the transport of OPAs is different from the transport of sediments for two key reasons:

- The OPAs do not in general have the same shape as oil droplets, and their density could vary from that of oil to a density much greater than that of water (due to sediments). Thus, they move differently from both the oil droplets and the sediment particles; and
- OPAs can form based on interaction with the wash load of rivers and not with the suspended and bed loads. Therefore, traditional sediment transport, which predicts only suspended and bed loads, might not be able to capture the dynamics of OPA transport when the OPA is generated based on interaction of oil with the wash loads.

5.2.4 Transport in estuaries

Flow in estuaries is characterized by reversals in flow direction due to tides and by the presence of a freshwater-saltwater interface. Large estuaries could have waves and could be strongly affected by wind and by the Coriolis force, which would tend to push the water to one of the banks of the estuary (Fischer et al. 1979). Estuaries tend to have a salt water toe overlain by fresh water exiting to sea. Therefore, for oil spilled inland in the estuary, dispersed or dissolved oil moving seaward would upwell to the water surface when encountering the saltwater toe (Short 2015). Thus, previously dispersed oil could re-coalesce again. The interaction of fresh and saline waters in estuaries is important not only for physical

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transport of spilled oil, but also because mixing of these waters often results in the combination and settling of clay particles due to the suppression of the doubl e layer when fresh water approaches high salinity waters (Clark 1996).

5.2.5 Transport of oil within beaches

Transport of oil within beaches is not commonly modeled, as the free -phase oil washes out fast, and one ends up with residual amounts of oil trapped by capillarity and by organic content in sediments (Brost et al. 2000) . How ever, sand beaches contain little organic material. Neglecting the organic content in sediments, Etkin et al. (2008) developed a holding capacity of beaches based on the beach sediment type and the tide range using the subsurface flow model MARUN (Boufadel et al. 1999).

The persistence of oil in a beach depends on the type of beach sediments and the beach groundwater table. While addressing the *Exxon Valdez* oil spill (EVOS), Li and Boufadel (2010) found that oil persisted at locations where the groundwater table drops into a lower sediment lay er of low permeability, causing the entrapment of oil. Beach hydraulics was also found to affect the oxygen concentration in the oiled regions of the beach resulting in low oxygen concentrations (Boufadel et al. 2010) suspected to cause the persistence of the *Exxon Valdez* oil spilled in Prince William Sound. The mechanism is illustrated in **Figure 5.3**. The beach receives dissolved oxygen from the inland groundwater table and from the seawater during the high tide. As the tide ebbs, the oxygen propagates into the beach and g ets consumed by abiotic and biotic reactions. For the EVOS, it was found that the oxygen concentration at the oiled region is around 1.0 mg/L resulting in anoxic conditions, which prevented aerobic oil biodegradation.

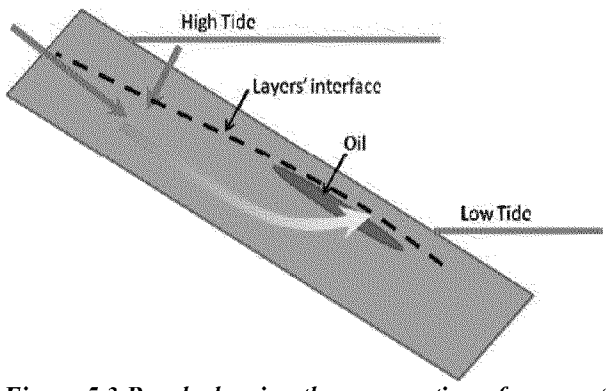

Figure 5.3 Beach showing the propagation of oxygen *(blue arrows) through the beaches of Prince William Sound, AK. Reprinted with permission from Boufadel, M.C., Sharifi, Y., Van Aken, B., Wrenn, B.A. and K. Lee. 2010. Nutrient and oxygen concentrations within the sediments of an Alaskan bea ch polluted with the Exxon Valdez oil spill. Environmental Science & Technology 44: 7418-7424. Copyright (2010) American Chemical Society.*

The role of nutrients in oil biodegradation has been noted since the 1970s (Atlas and Bartha 1972). But it gained a large visibility following the EVOS (Prince 1993; Bragg et al. 1994), where more than 50 tons of neat (i.e., solid) nutrients were dissolved a nd applied on to the beach surfaces. Wrenn et al. (1997) conducted tracer studies on a Delaware beach and found that the bulk of the nutrient plume persists in the beach for about 24 hours and thus re -application would be needed. This is understandable noti ng the hydraulics within the beach, as illustrated by **Figure 5.3**.

Tracer studies in Prince William Sound beaches were conducted by applying tracer s on the beach surface at low tide (Li and Boufadel 2011) and by releasing them deep into the beach (Boufadel and Bobo 2011; Boufadel et al. 2011) . In all cases, it was found that the applied solution move s slightly downward and landward during rising tides, and seaward during falling tides, confirming the numerical studies on these beaches (Guo et al. 2010, Xia et al. 2010).

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More recently, Geng et al. (2015) noted that sand beaches in the Gulf of Mexico could have different limiting factors depending on the location within the beach: nutrients could be limiting in the shallow zone of the upper intertidal zone, while oxygen could be limiting in the deep area of the mid -intertidal zone.

Bioremediation has been considered one of the major non-intrusive technique for mitigating the impact of oil on shoreline ecology (Pritchard et al. 1992) . The addition of nutrients is known as biostimulation (discussed in greater detail in Chapter 6) , while the addition of microorganisms is known as bioaugmentation (Zhu et al. 2001) . Prince et al. (1994) reported high degrees of success when nutrients were added to oil polluted beaches to remediate the EVOS. Venosa et al. (1996) conducted a bioremediation study on an experimental Delaware beach intentionally treated with oil in quintuplicate plots, and found that addition o f nutrients to concentrations larger than 2.0 mg/L in the interstitial pore water enhanced oil biodegradation several -fold. However, they noted that the addition of a microbial culture grown on site in drums exerted no further enhancement beyond simple nut rient addition. Later, Li et al. (2007) developed a nutrient delivery technique that applies to most beaches, which consists of applying the nutrient solution onto the beach surface following the falling high tide and to rely on the beach water table to carry the applied solution to the oil-contaminated zone.

5.2.6 Floodplains

An oil spill that occurs during a flood would result in oil deposition onto the floodplain, whose hydraulics is more or less disconnected from that of the river under normal conditions. This causes the oil to deposit in patches that are disconnected, especially if the spill occurred during high vegetation seasons. Therefore, the fate of oil in the floodplain could be greatly affected by photooxidation, some of whose products are soluble in water. The biodegr adation of oil in the floodplain could also occur, provided the sediments are sufficiently coarse to allow for oxygen replenishment from the atmosphere and also that the nutrient concentrations are not limiting.

5.2.7 Wetlands

In general, oil does not penetrate much into wetlands due to the fine texture of wetland sediment. However, channels and macropores due to bioturbation would constitute pathways to deeper penetration of oil. In addition, oil tends to adsorb to plants and plant roots. But in general the bulk of the oil gets trapped at the edge of the wetlands through adsorption to vegetation.

5.2.8 Underwater release

The release of oil under water due to a pipe rupture or well blowout causes the formation of a jet where the spewing oil moves in the water column due to the momentum. Eventually, the buoyancy of the oil starts to play a role, and the oil released at and beyond that location forms a plume. This is what happened in the DWH. However, for practical purposes one could refer to the oil release as a jet/plume. Upon rising through the water column, the vertical plume tends to get bent due to horizontal currents commonly present in waterbodies. Models that simulate such releases include the CDOG model (Zheng et al. 2003) and the Deepblow (Johansen 2000).

Jets/plumes generate a wide range of droplets whose transport and fate depends on the size. Small droplets tend to remain wit hin the plume even when it is bent by horizontal currents. However, large droplets rise rapidly and could separate from the plume bent by horizontal currents (**Figures 5.1 and 5.2**). Indeed, that was observed in the DWH spill, where the transport time of large droplets (size of millimeters) to the surface was on the order of hours, while that of small droplets (<70 μm) was on the order of months (Kujawinski et al. 2011). For this reason, the DSD from a pipe rupture is of great interest in terms of predicting where the droplets and plume will move. Unfortunately, the models CDOG and

DEEPBLOW do not generate the DSD; rather, they require it as input. Johansen et al. (2013) developed a formula for predicting the D_{50} (median diameter) from an underwater blowout based on the orifice diameter and the properties of the oil. Their model was calibrated to experimental data from vertical jets whose orifice was 1 to 5 mm (Brandvik et al. 2013), and one case where the orifice diamet er was 12 cm (Johansen et al. 2001). Using their model to obtain the D_{50}, the DSD would be obtained in the aftermath by assuming either a Rosin Ramler distribution or a lognormal distribution. The VDROP model combined with jet correlations resulted in the model VDROP -J, which provides the dropl et size distribution from jets/plumes (**Figure 5.4a**). The model was used to predict the DSD from the DWH in Zhao et al. (2015) (**Figure 5. 4b**). Zhao et al. (2014 b) argued that there is no theoretical reason for the existence of a unimodal DSD (Droplet Size Distribution), as required when using the Rosin Ramler or the lognormal distribution.

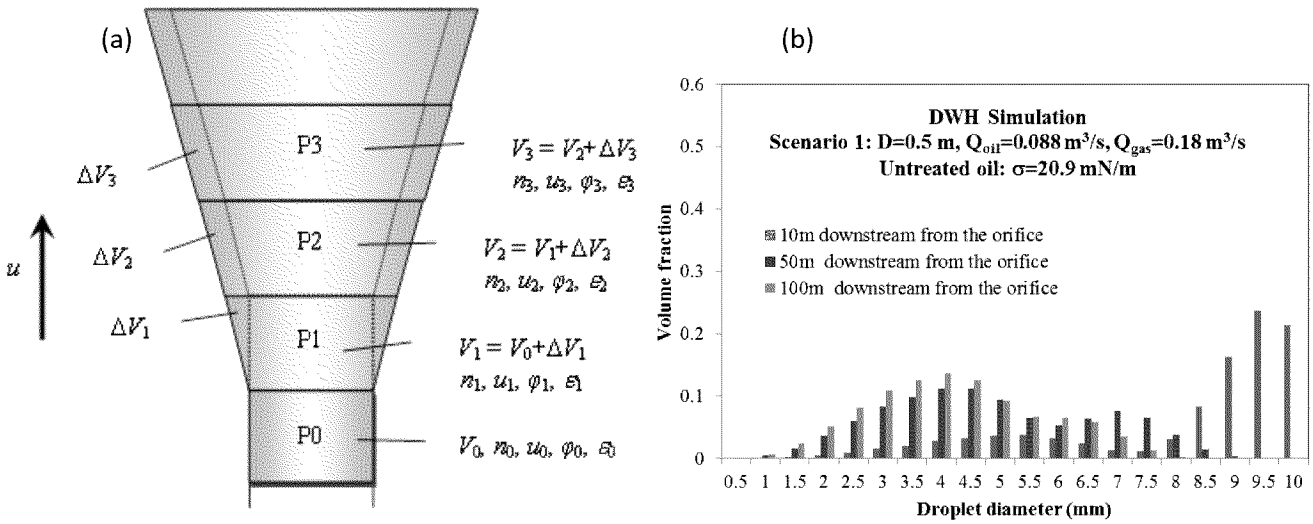

***Figure 5. 4 VDROP-J model**: a) illustration of the fluid parcel moving along the jet trajectory, VDROP model is running for each fluid parcel location to obtain DSD along the plume. Of interest are the following quantities: The number of droplets per unit volume "n", the stream velocity "u", the holdup φ (ratio of oil mass to total mass), and the energy dissipation rate ε, which vary as the oil moves away from the orifice; b) simulation results for DWH blowout[4].*

5.2.9 Ice

Lee et al. (2011) provided a thorough review of the behavio ur of oil in ice-infested waters in the Arctic. The findings of that report are summarized herein.

An obvious effect ice has on oil is to decrease the oil temperature causing the vi scosity of the oil to increase. However, weathering processes (addressed above) c ould still occur. In particular, for oil pooled between blocks of ice, evaporation becomes the dominant weathering mechanism. Also high viscosity causes the oil to spread less on water and to become less amenable to physical or chemical dispersion (Fingas and Hollebone 2003). Oil spilled directly on ice would tend to spread less than on water as the ice roughness would resist the spreading more than water (ITG 1983).

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[4] *Images reprinted from:*

a) Zhao, L., Torlapati, J., Boufadel, M.C., King, T., Robinson, B. and K. Lee. 2014b. VDROP: A comprehensive model for droplet formation of oils and gases in liquids -Incorporation of the interfacial tension and droplet viscosity. Ch emical Engineering Journal 253: 93-106. Copyright Elsevier (2014).

b) Zhao, L., Boufadel, M. C., Adams, E.E., Socolofsky, S., King, T. and K. Lee. 2015. Simulation of scenarios of oil droplet formation from the Deepwater Horizon blowout. Marine Pollution Bulletin. Accepted. Copyright Elsevier (2015).

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Oil spilled on ice can become encapsulated within the ice structure in winter conditions when new ice is being formed. In some case s, this process can happen rapidly (within 18 to 72 hours). In salt water, the formation of ice is accompanied by the release of salt, resulting in 'brine channels' and 'brine pools' (see Figure 2.10), which contain highly conc entrated solutions of sea salt. The channels provide pathways for oil migration into the water beneath the ice, and the pools provide mini -reservoirs for oil accumulation between the ice blocks (Bobra and Fingas 1986).

In terms of transport, studies have s hown that as long as the ice coverage is less than 30% of the water surface, the oil behaves more or less independen tly from ice. However, as the ice coverage exceeds 30%, the oil is found to drift with ice (Ross and Dickins 1987; Venkatesh et al. 1990). In that case, one would need to use ice models to track the movement of oil. Recently, major efforts have been placed on remote sensing, namely using satellites to detect the presence and thi ckness of oil under the ice (Dickins et al. 2008; Dickins et al. 2011).

5.3 Research Recommendations in Priority Order

- Better models need to be developed , using basic properties of waves, for the prediction of oil dispersion by waves, especially for high viscosity oils.
- Models for blowouts need to generate the oil droplet size distribution (DSD) and not only the underwater trajectory of oil. These would need to account for the effect of gas on the release. Currently, the only model that can generate the DSD is the model VDROP-J (Zhao et al. 2014b), which covers only the near-field.
- Predictive models are needed for the formation and fate of OPAs in freshwater and saltwater environments. Existing models are phenomenological; they reduce the data but cannot make predictions.
- Models for the biodegradation of oil droplets need to be validated by comparison wit h data obtained from controlled experiments, especially when dispersants are used (Venosa and Holder 2007).

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CHAPTER 6: REVIEW OF SPILL RESPONSE OPTIONS

Abstract

Chapter 6 discusses the evolving nature of oil spill response. The implementation and effectiveness of the various oil spill response options available are influenced by a variety of factors, such as oil types and properties (presented in Chapter 2), environmental and ecological conditions in the vicinity of the spill site (e.g., weather, wave, wind, daylight, ice conditions, visibility and sensitivity and distribution of species) (introduced in Chapter 3), technical, logistical and financial factors (such as responders' knowledge and skills, availability of personnel and equipment, time constraints, regulatory approvals, health and safety criteria, cost and economic impacts), and other constraints (e.g., community engagement). Some of those conditions and factors are temporally dynamic and interactive and should be comprehensively considered in oil spill contingency planning and response decision-making.

This chapter is divided into four main topical areas, namely: 1) natural processes; 2) physical response methods; 3) biological and chemical methods; and 4) factors affecting spill response and cleanup effectiveness. Under each main section, up to eight subsections present in detail the strengths and weaknesses of each response strategy to provide the reader with a clear understanding of how to implement each method based on influencing factors, such as how each method is affected by the environment and the type of oil being removed, and how each method is advocated or opposed for use based on its known effectiveness in different environments or ecosystems (depending on factors such as oil types, weather, wave impingement, ice conditions, daylight and visibility, ecological considerations and technical and economic conditions). Section 6.1 describes natural processes, such as natural attenuation, evaporation and photooxidation. Section 6.2 presents descriptions of physical cleanup processes, including containment and recovery, sorption, shoreline types, vegetation cutting, removal of oiled sediment, oiled sediment reworking, physical dispersion (influence of mineral fines) and in situ burning. Section 6.3 is concerned with biological and chemical strategies, such as bioremediation, phytoremediation, chemical dispersion, surface washing agents, solidifiers, herding agents and debris and detritus removal. Section 6.4 presents a discussion of the factors that affect spill response and cleanup effectiveness, such as oil types and properties, environmental and ecological factors (weather, wave height, ice conditions, daylight and ecological factors, including fish, invertebrate and wildlife mortality), and technical and economic factors impacting effectiveness.

Research recommendations for spill response needs and knowledge gaps are identified in priority order in Section 6.5. The underlying theme of the recommendations is the need to conduct controlled field studies to advance spill response strategies, especially for subsurface blowouts, Arctic oil spills, and freshwater shorelines. Most of what we know about oil spill response technologies has been developed from laboratory studies, mesocosm test systems and case studies (with limited controls and treatment replication). The stage for these studies should be primarily the Arctic, which has already been put forth in the needs from Chapter 3, because that is where our spill response knowledge is most lacking. We need to develop sufficient knowledge in understanding the fate, behaviour and impacts of different types of oils (in varying weathered and treatment states) in snow or ice conditions. Controlled field studies will help close this gap without significant negative impact on the environment if done properly. Furthermore, there is a need to improve our understanding of what to do about anaerobic biodegradation in sediments and to advance emerging technologies like bioventing, air sparging and/or dissolved air flotation to incorporate oxygen temporarily into the anaerobic zone of sediments and stimulate aerobic biodegradation of contaminating hydrocarbons. The results of such studies may aid in the cleanup of benthic environments contaminated by sunken oil from accidental spills or deep sea blowouts. Finally, regarding dispersant use, although some researchers have concluded that deep sea dispersant applications were successful during the Deepwater Horizon spill, it remains unclear to what extent the oil was dispersed physically due to the high velocity of oil emerging from the well into the water or chemically due to mixing with applied dispersant. This question needs to be answered by controlled experiments to reduce the uncertainty of this

treatment strategy. Questions about the persistence of dispersed oil in both the deep sea and on the sea surface need to be addressed with controlled empirical experiments.

Introduction

Oil spill response can be laborious, expensive and fraught with conflicting priorities. The implementation and effectiveness of the various oil spill response options available are influenced by a variety of factors, such as oil types and properties, environmental and ecological conditions in the vicinity of the spill site (e.g., weather, wave, wind, daylight, ice conditions, visibility and sensitivity and distribution of species), technical, logistical and financial factors (such as responders' knowledge and skills, availability of personnel and equipment, time constraints, regulatory approvals, health and safety criteria, cost and economic impacts), and other constraints (e.g., community engagement). Some of those conditions and factors are temporally dynamic and interactive and should be comprehensively considered in oil spill contingency planning and response decision-making.

> Spill response strategies are varied and greatly affected by a variety of factors, such as the type of oil (crude, refined, dilbit, etc.), the characteristics and remoteness of the spill site (water or ice, terrestrial, marine or freshwater shoreline, riparian zone, arctic, permafrost, etc.) and even political considerations.

Many of the approaches and technologies that have been used for treating oil spills were developed for marine shorelines, on-water-spills, and freshwater environments. These processes have been reviewed and described extensively in a number of technical documents, such as: Doerffer (1992), NOAA (1992), NOAA and API (1994), U.S. EPA (1999), NRC (2014), Canadian Coast Guard (2005), IMO (2005), Fingas (2011) and numerous ASTM manuals, monographs and data series (http://www.astm.org/DIGITAL_LIBRARY/MNL/mnltocall.htm). The most commonly used spill response cleanup options (**Table 6.1**) are briefly described in the following text. The rest of this chapter summarizes these technologies in more detail.

Table 6.1. Spill response cleanup options with chapter section designations that discuss each

Category of Response Options	Example Technology (Section)
Natural processes	Monitored natural attenuation (MNA) (6.1.1) Evaporation (6.1.2) Photooxidation (6.1.3)
Physical processes	Containment and recovery (6.2.1) Sorbents (6.2.2) Removal from shorelines (6.2.3) Vegetation cutting (6.2.4) Removal of oiled sediment (6.2.5) Oiled Sediment reworking (6.2.6) Physical dispersion (6.2.7) *In situ* burning (6.2.8)
Chemical and biological processes	Bioremediation (6.3.1) Phytoremediation (6.3.2) Chemical dispersion (6.3.3) Surface washing/flushing (6.3.4) Solidifiers (6.3.5) Herding (6.3.6) Debris removal (6.3.7)

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6.1 Natural Processes

6.1.1 Monitored Natural Attenuation (MNA)

In this section, monitored natural attenuation (MNA) is discussed in the context of circumstances that would indicate it to be the method of choice for a given spill response. Examples of such cases include spills at remote or inaccessible locations when natural removal rates are expected to be rapid, or spills in sensitive wetland sites where cleanup actions by human intervention (e.g., trampling) may cause more harm than good. Since some natural weathering processes are slow (Figure 2.4), the monitoring component of MNA is essential to verify the loss of residual hydrocarbons and the recovery of impacted plant, animal and other species essential for an ecologically healthy habitat.

> MNA or natural recovery is an option that should always be considered when determining which response option should be implemented. It may be the best alternative especially in certain situations where human intervention could result in more harm than good. MNA is essentially a no direct response option that depends on nature to degrade the oil. For some spills, such as those that have contaminated highly sensitive wetland and salt marsh environments that serve as habitats for endangered species occupying several trophic levels, it is probably more cost-effective and ecologically sound to let the site recover naturally than attempt to intervene.

A primary mechanism of MNA is biodegradation. This natural process is particularly important in removing the non- and semi-volatile components of oil from the environment. The ultimate fate of spilled hydrocarbons in the environment is dependent on the role indigenous microbial communities play in the fate of the majority of hydrocarbons entering the biosphere (Leahy and Colwell 1990; Prince et al. 2010; Atlas and Hazen 2011). This is a relatively slow process and may require weeks, months or even years for microorganisms to degrade a significant fraction of oil, particularly if other factors (discussed in Chapter 3) are limiting. In recent years, with the advent of genomics and metagenomics research activities, a shift has occurred in our understanding of the ecological significance of microbial adaptation in response to oil spills that identify the change in structure and function of whole microbial communities in previously unknown ways (Macnaughton et al. 1999). Pre-genomics ecological studies relied on the use of microbial isolates to guide our understanding. In contrast to the results of studies of isolates that represented only a small percentage of the community structure developing in or near the spill site, recent advances in genomic studies have shown that microbial activity in low temperature and deep sea conditions may be quite robust (Atlas and Hazen 2011; Valentine et al. 2015).

6.1.2 Evaporation

Evaporation is an important natural cleansing process during the early stages of an oil spill, as the lighter weight, higher vapour pressure components in oil are removed rapidly when exposed to the atmosphere (Chapter 2). Depending on the composition and mass of the oil spilled, up to 50% of the more toxic, lighter weight components of oil may evaporate within the first 12 to 24 hours following a spill (U.S. EPA 1999).

6.1.3 Photooxidation

Photooxidation occurs when oxygen under sunlight reacts oxidatively with aromatic oil components (see Chapter 2; NRC 2003). Maki et al. (2001) reported that when Arabian light crude oil was first biodegraded and then exposed to sunlight, a substantial decline in the aromatics fraction with a concomitant increase in the resins and asphaltene fractions was observed. The oxygen content in the oil and thus the polarity increased as the irradiation was prolonged, and the bioavailability of the biodegraded

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Photooxidation aids in the removal of dissolved hydrocarbons in water. Aliphatic and aromatic hydrocarbons are oxidized photochemically to oxygenated, more polar compounds that are more soluble than the parent compounds. Photooxidation may also result in higher molecular weight products through condensation reactions, ultimately leading to tar and gum residues.

oil increased by the photooxidation. *Photooxidation changes the physico-chemical properties of oil and its related components with the oxygenated forms being more polar, increasing the toxicity of the oil (Wang and Fingas 2003). The photooxidation products, while toxic, would not likely occur at toxic concentrations in water. However, evidence has shown that photooxidation within the tissues of transparent aquatic species causes rapid mortality (Chapter 4).* Thus, photooxidation may either produce potentially higher biodegradation rates due to increased hydrocarbon bioavailability or potentially lower rates due to toxic byproducts, depending on the oil composition and species sensitivity.

6.2 Physical Response Methods

6.2.1 Containment and recovery

Booms and skimmers are usually the first response method to an oil spill on water. It has met with varying degrees of success, mostly limited, depending on weather and sea state conditions.

According to Etkin and Tebeau (2003), a functional effectiveness of 10-30% can often be realized using mechanical recovery, with levels of 50% and greater being achieved on occasion (78% in the *Julie N* spill, up to 95% in others, such as the 2007 Burnaby syncrude spill in British Columbia [Crosby et al., 2013]). During a spill response, the threat to sensitive near-shore/coastal habitats by a spreading oil slick can be mitigated by deploying booms to trap and concentrate the advancing slick so that the oil can be physically recovered from the surface of the water by specially equipped boats or other devices (skimmers). Booms are floating, physical barriers made of plastic (usually marine grade PVC or urethane with a molded polyethylene shell), metal (for *in situ* burning [ISB]), sorbent fabrics or other materials having a cylindrical float at the top with a weighted bottom skirt under the water, which slow and trap or contain the spread of oil (NOAA 2015). Typical photos of boom deployment are shown in **Figures 6.1** and **6.2**.

Booms are commonly placed across a narrow entrance to a larger mass of water so that oil is prevented from passing through into marshland or other sensitive habitats. Deflection booms are used to deflect oil away from sensitive locations, such as shellfish beds or beaches used by birds as nesting habitats. Fire booms, discussed in more detail in Section 6.2.10, are used to collect oil from a slick and concentrate it into a thickness suitable for burning. Booms for ISB are typically fabricated with floating metal cylinders at the top that support a boom and skirt constructed with fire-resistant materials.

Although booms can be used in calm water (e.g. for streams, canals, ponds, lakes), open water (for harbours and open ocean conditions) and some fast water environments (for rivers, streams, estuaries and moving water lakes), their effective operational range is limited by rough weather and winds that induce strong currents and breaking waves.

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Figure 6.1 Oil containment boom deployed in a saltwater application. *Source: NOAA*

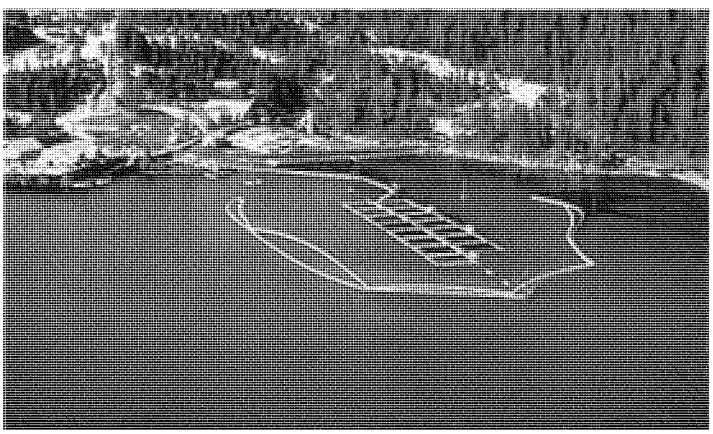

Figure 6.2 Boom surrounding a set of floating pens at a salmon hatchery in Prince William Sound, AK, to protect the pens from oil spilled from the Exxon Valdez tanker. *Source: NOAA*

Sorbent booms, also known as 'spaghetti' or 'sausage' booms, are used for collecting thick oil in flowing waters. They are composed of absorbent strips in the interior of the boom contained by open-cell netting. This type of boom allows oil to penetrate more deeply into the interior, which increases absorbency and helps prevent spreading from the spill site. Importantly, the Oil Budget Calculator study during the Deepwater Horizon (DWH) response (Federal Interagency Solutions Group (FISG) 2010) showed that despite almost ideal weather conditions and substantial logistical support, physical recovery accounted for only about 4% of the cleanup due to the large oil encounter rate from the continuous oil release to an open water environment over a period of several months. This estimate was later modified to 4-10% (Fingas 2013c) due to a calculation error by the FISG.

Skimmers are boats and other devices (e.g. weirs) designed to recover oil from the sea surface (e.g., **Figure 6.3 and 6.4**). Each type of skimmer has its own characteristics and limitations in dealing with factors such as viscosity, sea state and debris. Typically, two boats will tow a collection boom to concentrate the oil to facilitate its recovery. High waves may compromise the ability of boom containment and skimmers to remain in contact with the oil. Devices that have a small mass, allowing them to follow wave movements, should be part of the spill response inventory.

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Figure 6.3 A ship skims oil spilled after the Deepwater Horizon/BP well blowout in the Gulf of Mexico in April, 2010*. Source: NOAA*

Figure 6.4 Skimming the Delaware River in Philadelphia following the M/T Athos I spill on November 26, 2004. *Source: NOAA.*

Weir skimmers use a different oil collection method from those described previously. The skimmers float closer to the surface slick where the weir collects the oil while still allowing water to flow through. Some weir skimmers use a flotation system featuring four evenly spaced floats that help the weir adapt to changing water levels. This type of skimmer may be connected to a pump or vacuum system for rapid separation and removal of the floating oil. Vortex skimmers use an induced eddy to separate the oil from the water based on their differences in density.

> Skimmer systems such as weir skimmers may be more effective for high-viscosity oils, such as diluted bitumen, whereas skimmers with oil-attracting coatings may be more appropriate for conventional crude oil spills. Weir skimmers entrain large quantities of water, which enables them to cope with higher viscosity oils.

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According to Wadsworth (1995), oil viscosity affects the efficiency of virtually all recovery devices. Recovery is affected by the tendency of many oils to form water-in-oil emulsions, whereby, in addition to making the oil considerably more viscous, the volume of material is increased by three- to four-fold. The kinematic viscosity[1] may increase to as much as 100,000 cSt as a result of emulsion formation. The injection of demulsifying agents[2] can be used to mitigate this problem and minimize the storage volume required for recovered oil. Conventional weir, vortex, oleophilic rope and disc devices are usually limited to use with oils having a maximum viscosity of 10,000 cSt, whereas screw, belt and air conveyor devices will often perform well with oil of viscosities up to 100,000 cSt.

Physical recovery is limited by high wave action, leading to the loss of oil from containment booms either through oil splashing over the booms or to poor wave-following characteristics of the booms, resulting in bridging between crests. Skimmers are also subject to limitations from wave action as failure to remain in contact with the oil often results in the uptake of large quantities of water. In addition, turbulence can lead to loss of oil under the skimmer. Thus, a recovery device should have a small mass to enable faithful following of wave movements.

For oils that do not flow readily, such as heavy crudes and refined products, and ambient temperature below the pour point, the oil will behave as if it were a solid and thus resist recovery. If conditions are right, oil tends to form oil-in-water emulsions, increasing viscosity and volume of the emulsified material. The recovery efficiency, defined as the relationship between the quantity of oil relative to water in the collected material, is an important determinant of a system's overall performance. Some oleophilic systems recover small quantities of water but are less effective at high oil viscosities. These systems are best used with smaller vessels that are more effective in restricted water depths. Non-selective systems, such as weir skimmers, entrain larger quantities of water, and this property often enables them to deal with higher viscosity oils, such as dilbit or refined fuel oil.

6.2.2 Sorbents

This subsection is adapted from Region 10 Regional Response Team and the Northwest Area Committee (2015). The purpose of using sorbents as a spill response tool is to aid in the cleanup and recovery of oil-contaminated shorelines, sensitive habitats such as wetlands and salt marshes, and small spills on water. The primary uses of sorbents include the following (adapted from Merlin and Le Guerroué 2009):

- Containment and recovery by rapid deployment in coastal areas, ports and harbours, estuaries and on rivers;
- Containment of slicks in association with a standard boom (to improve watertight seal);
- Protection of areas difficult to clean (riprap, reed beds, mangroves, etc.);
- Immobilization or recovery of floating pollutants on lakes or in stagnant waters;
- Rapid application on terrestrial spills or ground surfaces to prevent or at least reduce infiltration of the pollutant to the substrate;
- Sorption of leaks below a recovery worksite;
- Sorption of effluents from cleanup of rocks, structures and embankments;
- Sorption by filtration of pollutants suspended in the water column (water intakes, rivers);
- Cleanup or decontamination of personnel and equipment on cleanup sites; and
- Lining and protection of pathways.

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[1]*Kinematic viscosity is the ratio of dynamic viscosity to density. The SI unit of dynamic or absolute viscosity is centipoise (cP) or mPa·s, and for water cP and cSt are equivalent. The kinematic viscosity of any fluid in cSt (centistokes) is the dynamic viscosity divided by the density of the fluid (in our case the oil). So, for oil, the kinematic viscosity will not be equivalent to the dynamic viscosity because of the division by oil density.*
[2]*Demulsifiers are a class of chemicals used to separate emulsions, e.g., water in oil, into oil and water phases.*

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An adsorbent takes up significantly less volume of liquid than an absorbent (Brinkman 1998). In Canada, sorbent pads and booms are frequently used in spill response. However, loose organic materials, such as peat, clay, straw, feathers, etc., are less prevalent because of the creation of more oily waste that must be burned or disposed to landfills. The preference is to collect bulk oil.

6.2.2.1 Selection of sorbents for floating oil spills on water

For fluid oil on open water with no current, bulk, pads and pillows should be used for containment. On open water with current and smooth-surfaced embankment, pads, pillows or sorbent booms with ballasted skirt should be used. On rough-surfaced embankments, either roll or sorbent booms with ballasted skirt should be used. For slow current (< 0.2 m/s) and small quantities of oil, sorbent rolls are used. For stronger current in a larger waterbody, standing floating booms or sorbent booms are used, but if these fail, sorbent rolls reinforced with rope are used instead.

For viscous oil, such as cold heavy fuel oil, dilbit or weathered emulsified crude oil, in open water with no current, standard floating booms or sorbent booms with ballasted skirt are used for containment or bulk and mops for recovery. In open water with current, rolls or sorbent booms with ballasted skirt or even mops are used if the current is slow. For deflection or recovery in faster current, sorbent booms with ballasted skirt or mops attached to a rope are used.

Recently, a natural plant-derived sorbent (sugarcane bagasse) was investigated in laboratory microcosms as a remediation strategy for low energy intertidal wetlands contaminated by crude oil (Chung et al. 2010). The results indicated that the use of this type of sorbent was beneficial not only in removing oil but also in preventing further contamination. This technique has potential to stimulate biodegradation by wicking oil out of contaminated anaerobic wetland subsurface to the aerobic zone on the surface where rapid biodegradation can take place. The bagasse, being biodegradable, would not need to be removed subsequent to treatment.

The American Society of Testing and Materials (ASTM) developed performance standards for adsorbents (Method F726) and absorbents (Method F716). The combination of the liquid with the solid causes the absorbent to swell, thus absorbing many times its volume. The liquid is no longer available for release and the rate of vapour release is reduced by five to six times that of adsorbents.

Three types of sorbents may be used in oil spill response: (1) products of mineral origin, such as expanded perlite or glass wool; (2) products of animal or vegetable origin, such as coarse sheep's wool, peat or cellulose peat or cellulose; and (3) synthetic products and organic polymers, such as polypropylene or polyurethane. Sorbents are selectively either hydrophobic, designed to recover non-polar pollutants (non-miscible with water), or hydrophilic, designed to recover polar products, such as water or materials soluble in water or non-polar products. Because these types of sorbents do not necessarily float on water but soak it up, they are used only on land.

Recommendation: Sorbents are widely used as a cleanup tool to combat oil spills in water and on shorelines. They have limitations, however. The need exists to determine if any advances have been or can be made to improve sorption as an effective response tool, especially in the use of biodegradable natural organic sorbents, such as bagasse that may possibly be left in place for biodegradation to take place. Significant amounts of debris and solid waste are generated when using sorbents for cleanup, so if natural sorbents can be developed that would sorb oil and then remain in place for ultimate cleanup by biodegradation, this would be a significant advancement in oil spill response.

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6.2.3 Oil removal from shorelines

Various beach washing techniques at ambient and high temperatures under both high and low pressure settings have been used as part of containment and recovery operations to treat different types of substrate contaminated by oil.

6.2.3.1 Sand beaches

When oil first arrives on a beach, deposits of fresh product must first be removed before cleanup can take place by any means. One technique used for free product removal is scraping and either bagging or trenching (**Figure 6.5**). Large volumes of bulk oil are removed quickly, without generating excessive waste, by scraping the free product directly from the sand surface. Rubber squeegees are used to manually concentrate the oil, which is then pushed into trenches dug parallel to the waterline. For long stretches of contaminated sandy beach, the oil is collected by vehicles equipped with scraper blades and dumped into temporary storage pits above the high tide line. The oil is collected by vacuum trucks for later treatment.

Figure 6.5 Workers clean a sandy beach using traditional methods of hand labour and bagging the oil and debris. *Source: NOAA*

6.2.3.2 Pebble or shingle beaches

These types of beaches are "armoured" with pebbles or small- to medium-sized cobbles as opposed to fine sand. They are steeply sloped because waves are able to course through the highly porous surface, which decreases the effect of backwash erosion and increases sediment formation. In these low energy beaches, vehicle access is usually possible, and bulk oil is either collected into pools onto the beach surface or flushed into man-made trenches. The recovered oil is then collected in vacuum trucks and either transferred to temporary holding tanks or taken away for treatment. *However, pebble beaches are difficult to clean thoroughly because oil tends to penetrate into the subsurface, sometimes as far as the clay/mud base. Thus, flushing into trenches requires later stages of cleanup, usually by pebble washing. In such cases, a better cleanup method is to flush the oil off the beach at high tide into offshore booms, where the oil can be collected by skimmers.*

6.2.3.3 Cobble beaches

These types of beaches are characterized by higher energy wave action impinging on the surface. Thus, surf washing (also known as berm relocation or oiled sediment reworking) is often used for treating

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cobbles that have been exposed to beached oil (Lunel et al. 1996; Lee et al. 1997; Lee et al. 2003). Such cobbles are placed in the high energy surf zone where they are scoured naturally by the wave action. Where removal of large oiled cobbles has been difficult, large pits lined with plastic have been dug into the beach where the cobbles are washed with an oil-releasing agent and water. Oil is then skimmed and the clean cobbles are returned to the beach surface. However, this method usually leaves the cobbles oil-stained and is not used in sensitive environments, such as salmon spawning streams, shellfish beds, nursery areas, etc., because of the potential adverse effects on those sensitive habitats from the increased oil runoff, sheening or siltation.

Recommendation: Berm relocation, surf washing and sediment reworking have been used to accelerate shoreline cleanup in the past. A comprehensive review is needed to determine quantitatively how successful these technologies have been and where such techniques are best applicable or not recommended. Testing protocols need to be developed to determine the effectiveness and secondary ecological impacts of this response option.

6.2.3.4 Boulder beaches

Boulder beaches are usually found in high wave energy environments where clasts of these large dimensions are released directly by erosion of bedrock, or where material is delivered to the shore zone by slope movements, such as rockfall. Oil stranded in these areas can become mixed with sand, forming an oil/sand coating on the rock surfaces. This coating can be removed relatively easily by surface scrubbing, brushing or wiping **(Figure 6.6)**. However, re-oiling often occurs in these types of areas due to the constant movement of sand during tidal cycles, resulting in a protracted cleanup. Thus, wiping is a highly labour intensive treatment technique. *Other techniques, which may be more cost-effective than wiping include high pressure cleaning with or without the use of surface washing agents. If the impinging wave energy is high enough, natural recovery via physical abrasion from the waves is probably a better option.*

Figure 6.6 Workers clean oiled boulders by hand. *Source: NOAA.*

6.2.3.5 Bedrock shoreline with no beach

This shoreline type is frequently found along the coastal areas of Canada and common in many Canadian lakes, including portions of the Great Lakes (e.g., **Figure 6.7**). Rock surfaces on marine shorelines are often densely populated with kelp (and associated biofilm). They can have a thriving tide pool community and therefore should not be regarded as 'barren'. Furthermore, wet kelp may reduce oil adhesion to rock surfaces. **Figure 6.8** shows workers cleaning oil spilled following a collision involving crude oil tanker *Eagle Otome* in Port Arthur, TX, on January 2010. Under inclement weather conditions following a spill,

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waves may deposit oil up on the rocks, which may have a lichen/moss coating or be inhabited by other vegetation, such as small willows or conifers that cling to the cracks in the rock.

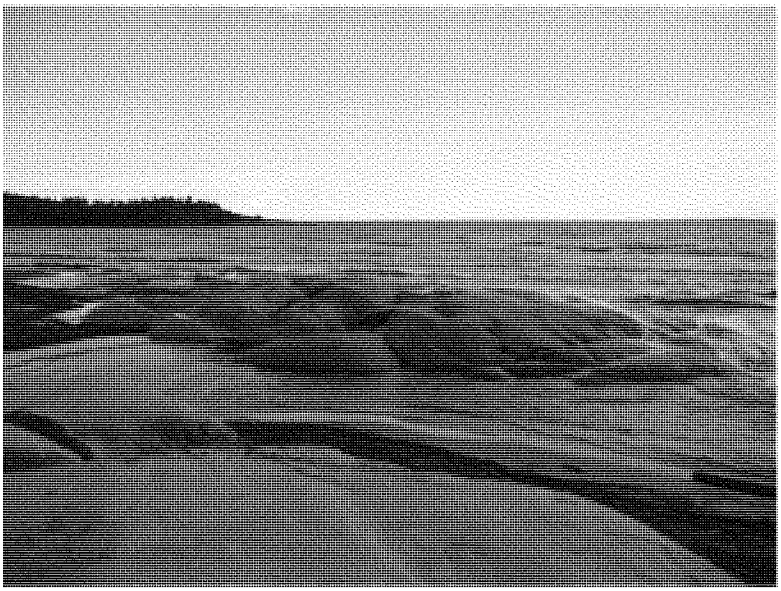

Figure 6.7 North shore of Lake Superior. *Source: NOAA.*

Figure 6.8 Workers cleaning oil spilled following a collision involving crude oil tanker Eagle Otome in Port Arthur, TX, on January 2010. *Source: NOAA.*

6.2.3.6 Low pressure washing or sediment flushing of a variety of beach types

Flooding a beach (**Figure 6.9**) with seawater can be used to flush fluid oils and oily debris from different shoreline types, particularly in sensitive areas (IMO 2005). This approach is less damaging to flora and fauna than most other methods. Booms are used to contain the flushed oil for skimming and removal.

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Figure 6.9 Flushing out oil buried in a sand beach using low pressure water supplied through lances and perforated pipes. *Photo Credit: ITOPF; http://www.itopf.com/knowledge-resources/documents-guides/response-techniques/shoreline-clean-up-and-response/*

6.2.4 Vegetation cutting

Vegetation is cut to remove oil-contaminated vegetation from shorelines, wetlands or salt marshes **(Figure 6.10)** to prevent contamination of wildlife or remobilization of trapped oil. This strategy is based on the removal of oil from impacted environments by the pruning and collection of the upper parts of oiled plants for waste disposal. The cut vegetation is then collected and disposed. Any oil remaining near the roots or around the stems can be flushed out and recovered. Cutting can result in a temporary loss of habitat for small fish and invertebrates that dwell in those areas. Its feasibility depends strongly on the season in which the spill occurs. In general, winter cutting of dead standing vegetation has little effect on subsequent growth, but summer cutting could cause great damage to the regrowth of wetland plants and result in shoreline erosion. The use of cutting should also be avoided immediately prior to an anticipated rise in water levels because cutting followed by flooding could cut off necessary oxygen to plant roots (Pezeshki et al. 2000). The plants may or may not recover, depending on the type of oil spilled and differences in species and life stage sensitivity. This response method is generally used when large amounts of potentially mobile oil are trapped in the vegetation or when the risk of oiled vegetation contaminating wildlife is greater than the value of the vegetation that is to be cut. This response is conditionally recommended for sheltered, vegetated, low banks and marshes. See Chapter 4 for a discussion of the environmental impacts of vegetation cutting in the floodplain zone.

Vegetation cutting is also used for the removal of oil in the canopy of kelp beds, where thick layers of oil may adhere to kelp fronds or collect under the kelp canopy. This procedure is similar to shoreline vegetation cutting. The upper ~0.5 m of the kelp canopy are cut away by hand or with a mechanical kelp harvester and removed for disposal. Vegetation cutting is used when a large quantity of oil is trapped in the kelp canopy and the oil poses a risk to sensitive wildlife using the kelp habitat or when the remobilization of oil to other adjacent sensitive environments is likely to occur.

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Figure 6.10 Cleanup workers removing oiled vegetation during the Deepwater Horizon spill and using walk boards while cleaning to avoid causing further damage to the oiled marshes in Barataria Bay*. Source: NOAA.*

6.2.5 Removal of oiled sediment

This response technique is used to remove oiled surface sediments that cannot otherwise be cleaned *in situ*. Oiled sediment is removed by use of hand tools (**Figure 6.11)** or motorized equipment. Oiled sediment removal is restricted to the supratidal and upper intertidal areas to minimize disturbance of biological communities in the lower intertidal and subtidal zones. After removal, oiled sediments are transported for cleaning and/or disposal offsite. New sediments are not usually imported to replace those that were removed; however, a variation of this response that includes sediment replacement (described below) is used for beaches with low natural replenishment rates or high rates of erosion.

Physical removal of oiled sediment is frequently used on high amenity 'tourist' beaches (where pressure is high for rapid cleanup) and is most effective when a limited amount of oiled sediment must be removed. Close monitoring is required so that the quantity of sediment removed, siltation and the likelihood of erosion may be minimized in all cases. Such operations are generally restricted in fish spawning areas. Sensitive areas that are adjacent, and may be potentially affected by released oil sheens, must also be protected. During the DWH spill at the Grand Isle State Park in Louisiana, remediation operations included the excavation of the top few centimetres of contaminated sand for the removal of bulk oil by physical/chemical extraction at a temporary facility constructed near the spill site and its subsequent return to the beach **(Figure 6.12)** where the treated sediments were also surf washed for final polishing.

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Figure 6.11 Cleanup workers manually remove oil following the M/V Westchester spill in the Mississippi River near Empire, LO, on November 2000. *Source: NOAA.*

Figure 6.12 Oiled sand removal in Grand Isle State Park following the DWH spill. *Source: K.Lee, DFO.*

6.2.6 Oiled sediment reworking

The objective of this variation of oiled sediment removal is to rework oiled sediments to break up oil deposits, increase surface area, facilitate physical oil dispersion and mix oxygen into deep subsurface oil layers. This activity exposes the oil to natural removal processes and enhances the rate of oil biodegradation. Under this protocol, oiled beach sediments are rototilled or otherwise mechanically mixed with the use of heavy equipment or relocated from the upper intertidal area of the beach to the surf zone. This latter procedure is also known as surf washing or berm relocation (Section 6.2.3). Generally, sediment reworking is used on sand or gravel beaches where high erosion rates or low natural sediment replenishment rates are issues. Sediment reworking may also be used where remoteness or other logistical limitations make sediment removal unfeasible. Sediment reworking is not used on beaches near shellfish harvest or fish spawning areas because of the potential for release of oil or oiled sediments into these sensitive habitats. *Sediment reworking is conditionally recommended for: (1) sand beach; and (2) cobble and gravel beach habitats* (Lunel et al. 1996; Region 10 Regional Response Team and the Northwest Area Committee 2015).

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A new oil spill monitoring protocol called 'poling' was used in the Kalamazoo River spill in 2010 as a means of finding oil in the sediment. Poling involves disturbing the river sediment with a pole every 50 m or so to locate any remnants of oil residue remaining in the riverbed from the spill. Oil distribution in the sediments was estimated from the observation of oil sheens on the surface after disturbing the riverbed by poling. As a result of this survey method, identified areas with significant oil concentrations within the sediment were physically agitated by water jet washes as a means to facilitate residual oil release to the surface waters where it was physically recovered by skimming and the use of sorbent pads.

6.2.7 Physical dispersion (Oil mineral aggregates)

Suspended particulate matter within the water column, such as plankton and mineral fines, may form nuclei around which oil droplets may interact and adhere (Muschenheim and Lee 2003) to form oil-mineral aggregates (OMAs; Chapter 2). The subsequent ecological consequence of this process is dependent on the oil concentration and sedimentation rates. If the material is retained within the water column by physical oceanographic processes, oil toxicity may be reduced due to dilution of the residual oil by dispersion and enhanced oil biodegradation rates (as the process may enhance the surface area of the oil and nutrient availability; Figure 2.3). However, if the material rapidly sinks into the sediment and accumulates at elevated concentrations, biological impacts may result and the oil may degrade very slowly and thereby persist for long periods of time (see Chapter 3 for a discussion of anaerobic conditions) (especially if entrained within anaerobic sediments; Figure 2.3).

Several *in situ* remediation technologies, bioventing, air sparging and dissolved air flotation (DAF), developed over the last 20 years for the treatment of contaminated soils and groundwater have not been tried for remediating bottom sediments contaminated by sunken oil from spills. These technologies are characterized by injection of oxygen-rich air into the sediment to accelerate the aerobic biodegradation of oil in that anaerobic environment. Specifically, DAF achieves contaminant removal by dissolving air in the water or wastewater under pressure and then releasing the air at atmospheric pressure in a flotation tank or basin. The released air forms tiny bubbles that adhere to the suspended matter causing it to float to the surface of the water where it may then be removed by a skimming device.

Recommendation: Bioventing, sparging and DAF should be evaluated for their efficacy in treating and overcoming the adverse effects of sunken oil in the anaerobic zone of river or lake sediments. If successful, such techniques would greatly help in cleaning up benthic environments that have been contaminated by oil.

6.2.8 In Situ *burning*

ISB, a technique used to rapidly reduce the mass of oil spilled at a site, involves the controlled burning of oil that has spilled from a vessel or a facility at the location of the spill. ISB was successfully demonstrated by Environment Canada in a large-scale field experiment, the Newfoundland Offshore Burn Experiment (NOBE), on August 12, 1993. During each of two test burns, crude oil was poured into a towed U-shaped fireproof boom and ignited. The first test burn lasted for an hour and a half and the second for about an hour, with an average observed burning rate of 24 tonnes of oil per hour (Fingas et al. 1995a). ISB was conclusively validated as an operational physical oil spill response countermeasure following controlled burns near a swamp (**Figure 6.13**) and at sea (**Figure 6.14**) during the DWH incident.

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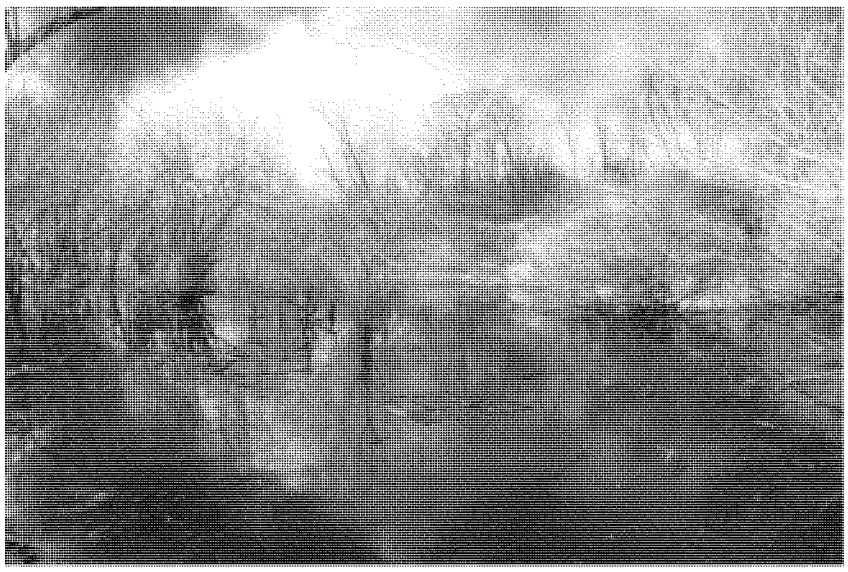

Figure 6.13 A view of one of the controlled burns to remove oil spilled in a wooded swamp outside of Baton Rouge, LA, on January 19, 2013. *Source: USCG.*

Figure 6.14 After the Deepwater Horizon oil spill in the Gulf of Mexico in April 2010, ISB was one of the techniques used to remove oil from the water. *Source: NOAA.*

During at-sea ISB operations, oil on the sea surface is captured within a boom towed by two boats in a U-configuration and ignited using a hand-held igniter or an igniter suspended from a helicopter. Under favourable conditions ISB is a fast, efficient and relatively simple way of removing spilled oil from the water to minimize the adverse effect of the floating oil on the environment. Furthermore, it greatly reduces the need for storage and disposal of the collected oil and the waste it generates. The burn will continue only as long as the oil slick is thick enough—usually about 2-3 mm. Burning oil results in residues (approximately 1-5% of the starting oil) from incompletely combusted oil and gaseous emissions into the atmosphere that have raised environmental concerns. This is particularly true if the residue sinks.

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Results of laboratory tests suggest the possibility that burn residues from ~40 to 60% of crude oils may sink. *However, whether results from laboratory tests can be extrapolated to large-scale spills is not known (NOAA 2015).*

To burn oil on the water, slicks must be thicker than 1-2 mm. During combustion the oil vapours ignite and burn, rather than the liquid itself. When the oil layer is thinner than ~1 mm, the heat is lost to the water, not enough vapours are released, and combustion ceases. Experiments have shown that ISB is best used under relatively calm conditions, as choppy seas may extinguish the fire altogether (ITOPF, http://www.itopf.com/knowledge-resources/documents-guides/response-techniques/in-situ-burning/). When the current is stronger than about one knot, the boom cannot contain the oil, which splashes above the boom or escapes beneath it. This resulting entrainment typically occurs when booms are deployed perpendicular to the water flow (API 2015). Water-in-oil emulsions of over 50% will preclude ISB of even light crudes or refined products, while much less than that is required for heavier crudes (NOAA 2015).

Studies of the emissions from ISB have shown fairly consistent results. According to Ferek et al. (1997), about 85 to 95% of the burned oil is converted to carbon dioxide and water, 5 to 15% of the oil is not burned efficiently and is converted to particulates, mostly soot, and the rest, 1-3%, comprises nitrogen dioxide, sulfur dioxide, carbon monoxide, polycyclic aromatic hydrocarbons, ketones, aldehydes and other combustion by-products. The 5-15% estimated conversion to particulates differs from the 1-5% reported above, suggesting that a number of factors are in play when making estimates of ISB conversions. No 'exotic' chemicals are formed. Rather, the burning of oil on water seems to be similar to burning the oil in a furnace or a car's internal combustion engine, with the exception that the burn is oxygen-starved and not very efficient, so that it generates ample amount of black soot particulates that absorb sunlight and create black smoke. ISB is not used if human populations are located near and downwind from the site. For oil spills occurring in ice-covered water or wetland or salt marsh environments, besides MNA, ISB may be the only spill response method available. ISB, however, should complement, not exclude, other means of spill response. ISB has been used successfully on numerous occasions when oil was trapped in ice or spilled into sensitive marshland.

Recommendation: ISB is fast gaining a foothold as a response technology for treating floating oil slicks. More research, including peer-reviewed literature, is needed to ascertain harmful emissions released to the environment during ISB, such as any incompletely combusted polyaromatic hydrocarbons (PAHs), toxic particulate matter, etc., and to determine the safety and effectiveness of ISB as an oil spill response tool.

6.3 Biological and Chemical Methods

6.3.1 Bioremediation

Many hydrocarbons (normal and cyclic alkanes, most monoaromatics and some PAHs; Table 2.1) are biodegradable under aerobic conditions, although many polar resins, most hopanes and asphaltenes are resistant to microbial attack. Thus, bioremediation is a viable response technique for cleaning up a shoreline contaminated by an oil spill, with the caution that light crude oils are much more easily

> Bioremediation is the exploitation of the ability of microorganisms to convert pollutants, such as petroleum hydrocarbons, into biomass, carbon dioxide, water and innocuous oxygenated end products. The oil serves as food for the microbes, providing energy and carbon for reproduction and growth. Biodegradation is the process microorganisms use to obtain that end. Thus, bioremediation is a human intervention, whereas biodegradation is a natural property of microorganisms.

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biodegraded than diluted bitumen and heavy refined products, such as fuel oils. The passage of time has allowed the evolution of numerous diverse microbes to utilize hydrocarbons as a source of carbon and energy for growth (Atlas and Hazen 2011). It should also be noted that many of the oil deposits around the world, especially those near the surface, are actually residues of millions of years of bacterial degradation of organic, mostly plant-based material.

Two primary approaches are used in bioremediation: *bioaugmentation* and *biostimulation*. Bioaugmentation is the addition of an exogenous supply of oil-degrading microorganisms (grown offsite or in the laboratory) to the oil-impacted environment to supplement (augment) the existing microbial populations and accelerate biodegradation. Biostimulation describes the addition of nutrients or other growth-limiting chemicals, such as electron acceptors, to accelerate biodegradation by the indigenous microbial communities that are already present. Both these approaches have been extensively studied in the laboratory and the field.

The rationale for bioaugmentation is that indigenous microbial populations may not be capable of degrading the wide range of potential substrates present in petroleum (Leahy and Colwell 1990), and that addition of selected, competent microbes would permit efficient biodegradation. Other conditions under which bioaugmentation may be considered are when the indigenous hydrocarbon-degrading population is low, the speed of decontamination is the primary factor or when seeding may reduce the lag period to start the bioremediation process (Forsyth et al. 1995). However, bioaugmentation has never been demonstrated with statistical rigour to accelerate biodegradation to any appreciable extent in field studies. McGenity et al. (2012) pointed out that the reasons for past failures of bioaugmentation have been: 1) use of a pure culture rather than a community; 2) focus on biodegrading strains only, without partner microbes; 3) microbes not adapted to the environment and unable to compete for nutrients with indigenous microbes; 4) inadequate dispersion or access to the pollutant; 5) lack of protection from indigenous grazers or predators (e.g., amoebae); and 6) other factors limiting biodegradation (nitrogen and phosphorus).

Except for item 6, these factors were not in play in the statistically rigorous field research conducted by Venosa et al. (1996) on the Delaware shoreline involving MNA, biostimulation and bioaugmentation. In the bioaugmentation treatment, they applied both nutrients and indigenous microorganisms grown on site in 55-gallon drums containing the same seawater and crude oil and grown on the same shoreline under the same environmental conditions as the beach. The study design included quintuplicate plots of all three treatments, and bioaugmentation was unable to further accelerate biodegradation beyond simple nutrient addition, which did achieve statistically significant biodegradation rate enhancement. *While inoculation doesn't seem to be an issue, as reported by Jacques et al. (2008), the future development of good microbial consortia having complementary biodegradative pathways and bio-surfactant production capabilities to enhance the bioavailability of residual hydrocarbons may someday make bioaugmentation feasible on shorelines.*

Biostimulation has been successful in a number of laboratory, mesocosm and field studies, making it the strategy of choice in the open environment because of the ubiquity of oil-degrading bacteria (Venosa et al. 1996; Lee 2000; Zhu et al. 2001, 2004). Microbial population density is generally not the primary limiting factor over the time scales required for bioremediation. When high concentrations of oil are suddenly introduced into the environment, the microbial communities are faced with a huge increase in food (carbon) without a concomitant increase in the nutrients they need to grow at the maximal or near maximal rate (i.e., nitrogen and phosphorus; Chapter 3). Thus, the rates of biodegradation will be necessarily slow since growth of the microbes will be limited by the amount of nitrogen and phosphorus available for the transformations. Hence, biostimulation is used to supplement the limited available

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nutrients that indigenous microbes need to grow rapidly on the introduced carbon. This is true regardless of the environment in which the oil spill occurred (shoreline, open water, wetland, land, etc.). Two comprehensive guidance documents of oil spill bioremediation covering all aspects of microbial interactions with spilled oil, published in 2001 and 2004 (Zhu et al. 2001, 2004), are still relevant today.

The Redfield Ratio (defined and discussed in Chapter 3) is the stoichiometric relationship among carbon, nitrogen and phosphorus in plankton. To meet the theoretical demand for nitrogen and phosphorus during biostimulation, approximately 150 mg of nitrogen and 30 mg of phosphorus are consumed in the conversion of 1,000 mg of hydrocarbon to cell material (Rosenberg and Ron 1996). Therefore, a commonly used field strategy is to add nutrients at concentrations that approach a stoichiometric ratio of C:N:P of 100:10:1, which is similar to the Redfield Ratio of 106:16:1. However, the practical use of these ratio-based theories remains a challenge. Particularly in marine shorelines, maintaining a certain C:N:P ratio is impossible because of the dynamic washout of nutrients resulting from the action of tides and waves. A more practical approach is to maintain the concentrations of the limiting nitrogen and phosphorus within the pore water at a range optimal for rapid biodegradation (Bragg et al. 1994; Venosa et al. 1996). Commonly used nutrients include quick release water-soluble, solid slow-release and oleophilic fertilizers. Each type has its advantages and limitations. Zhu et al. (2001, 2004) and Venosa and Zhu (2005) provide guidance on how to maintain these concentrations in the field. *Importantly, the optimum C:N:P ratio for anaerobic hydrocarbon biodegradation in situ has not yet been defined.*

Recommendation: There is a need to quantify the persistence of oil of all kinds during and after biodegradation and oil that has been chemically-dispersed in the environment following high-impact spills to help stimulate innovations in bioremediation approaches. This is especially pertinent to cold temperature and anaerobic environments.

6.3.1.1 Wetlands

Wetlands, salt marshes and mudflats are among the most sensitive and difficult ecosystems to clean.

Productivity in coastal marsh ecosystems is extremely rich, and such systems provide habitat and breeding grounds to many vital species. Use of heavy equipment and human trampling are not amenable to a viable response activity because these cleanup actions may cause more damage than the spill itself. If human presence is needed, wooden walkways (**Figure 6.10**) or boat-based activities should be used as much as possible. Effective response techniques for these sensitive environments may include less intrusive techniques, such as:

- Booms to contain and control the movement of floating oil at the edge of the wetland (Section 6.2.1);
- Sorbents for removal of the oil by adsorption onto oleophilic materials placed in the intertidal zone (Section 6.2.2);
- ISB (but on a stringently limited basis) (Section 6.2.8);
- Bioremediation (Section 6.3.1);
- Phytoremediation (Section 6.3.2); and
- Low pressure flushing with ambient seawater at pressures less than 200 kpa or 50 psi to the water edge followed by removal (Section 6.3.4).

Dispersants (Section 6.3.3) are unlikely to be appropriate for use near a coastal wetland or salt marsh due to the lack of mixing energy to aid in effective dispersion into the water column, limited water depth to effectively disperse the oil below toxicity threshold limits, and the unknown effect of either the dispersed oil or the dispersant itself on ecosystems in the area. *The use of dispersants in near-shore water could have short-term toxic effects on adjacent coastal habitats, such as subtidal animal communities. Direct spraying or contact of dispersants with wetland plants may also cause harmful effects on vegetation.*

6.3.2 *Phytoremediation*

> Phytoremediation has been defined as the use of green plants and their associated microorganisms in the soil to degrade, contain or render harmless environmental contaminants

Phytoremediation (**Figure 6.15**), which takes advantage of the mutually beneficial interactions between plants and microbes for environmental cleanup (Cunningham et al. 1996)., is emerging as a potentially cost-effective option for cleanup of soils or sediments contaminated with petroleum hydrocarbons (Frick et al. 1999). As summarized by Macek et al. (2000), the main advantages of phytoremediation include less disruption to the environment, potential to treat a diverse range of contaminants and high probability of public acceptance. Major concerns regarding this technology include dissolution and migration of contaminants and relatively slow rates of remediation.

Degradation can be accomplished by both plants and associated microorganisms in the rhizosphere. *Plants can stimulate the growth and metabolism of rhizosphere microorganisms by providing root exudates of carbon, enzymes, nutrients and oxygen, which can result in more than 100-fold increase in microbial counts (Macek et al. 2000).*

A principal factor involved in the enhanced degradation of hydrocarbons in soil is the 'rhizosphere effect[3]', i.e., an increase in the numbers and activity of soil microorganisms in the plant-root zone. The plant-root zone has a high surface area, which contributes to growth of oil-degrading microorganisms (and other microbes) in response to high concentrations of hydrocarbons present in the rhizosphere following oil incursion. In addition to elevated microbial community numbers, the contribution of plant-root extracellular enzymes to the biodegradation of organic pollutants in the rhizosphere can be considerable. This has been demonstrated by the findings of Gramss et al. (1999), who showed that the roots of some plants release enough oxidoreductase enzymes to take part in the oxidative degradation of certain soil constituents. Active involvement of plant peroxidase enzymes in the phytoremediation process has also been suggested by several other authors (Kraus et al. 1999; Criquet et al. 2000; Chroma et al. 2002), including PAH degradation (Muratova et al. 2009).

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[3]*The rhizosphere is the zone of soil or sediment surrounding plant roots and roothairs that is directly influenced by the living plant, e.g., exudates, moisture, nutrients, etc.*

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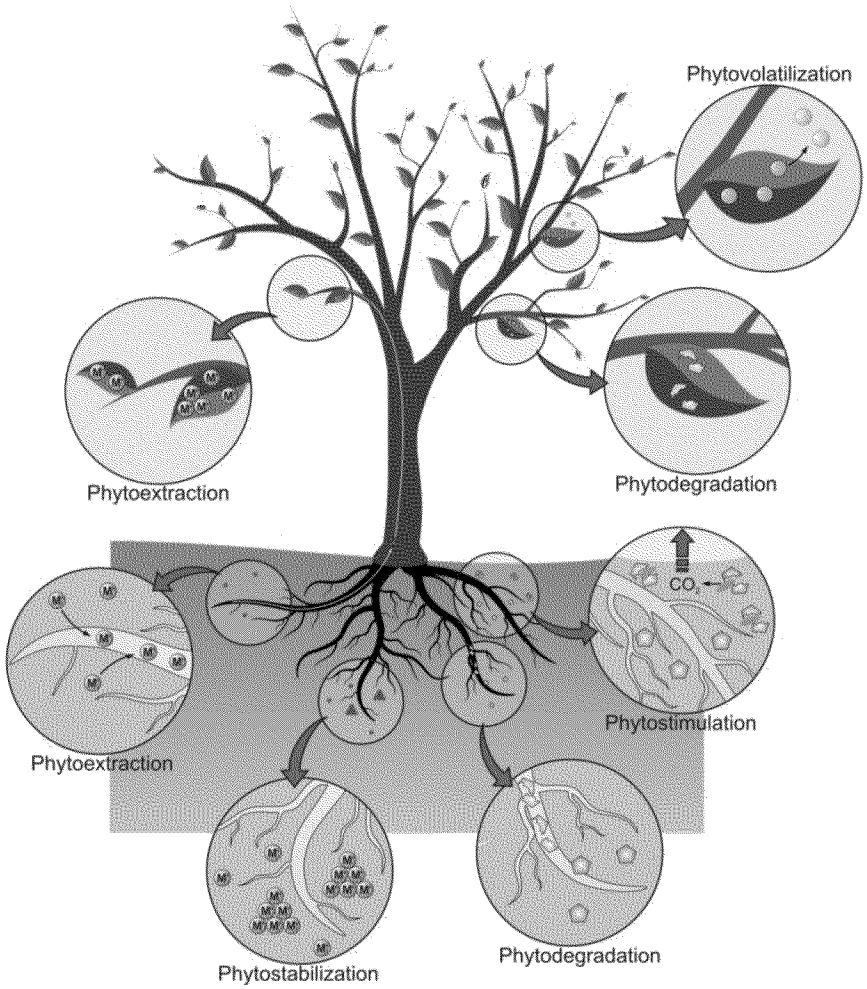

Figure 6.15 Schematic representation of phytoremediation activities. "M" indicates microbial cells. *Source: © 2014 Favas, P.J.C., Pratas, J., Varun, M., D'Souza, R., Paul, M.S. Published in Soriano, M.C. (ed.) Environmental Risk Assessment of Soil Contamination. InTech. Under CC BY 3.0 license. Available from: http://dx.doi.org/10.5772/57469*

Phytoremediation is mutually beneficial. The microbes can reduce the phytotoxicity of contaminants so that plants can grow in adverse soil conditions. Co-metabolism[4] may also play an important role in phytodegradation. Ferro et al. (1997) suggested that plant exudates might serve as co-metabolites in enhancing the biodegradation of the 4-ring PAH pyrene in the rhizosphere.

Other major mechanisms of phytoremediation include retention of petroleum hydrocarbons and their transfer from the soil to the atmosphere. Containment involves the accumulation of contaminants within the plants, adsorption of contaminants onto roots and binding of contaminants in the rhizosphere through enzymatic activities (Cunningham et al. 1996; Frick et al. 1999). Plants can also transport volatile petroleum hydrocarbons to the atmosphere through leaves and stems. However, these effects are less important than the rhizosphere effect during phytoremediation of petroleum hydrocarbons (Ferro et al. 1997).

[4]*Co-metabolism involves the partial oxidation of a chemical (e.g., hydrocarbon) by one organism (that receives no benefit from the metabolism), and the product is further metabolized by another organism that does receive benefit (e.g., can use the partially oxidized product for growth).*

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Several full-scale studies in the 1990s and early 2000s highlight what is known about salt marsh remediation. Lee and Levy (1991) conducted one of the first field trials on oil bioremediation in a salt marsh environment. The study involved periodic addition of water-soluble fertilizer granules (ammonium nitrate and triple super phosphate, providing the nitrogen and phosphorus) to enhance biodegradation of waxy crude oil in a salt marsh in Nova Scotia dominated by *Spartina alterniflora* (saltmarsh cordgrass). Results showed that the effectiveness of nutrient addition was related to oil concentration, i.e., effectiveness was high at 0.3% oil contamination by mass, but not at 3% oil contamination. Lack of effectiveness at the higher oil concentration was attributed to the penetration of the oil into the reduced soil layers where anaerobic conditions precluded rapid degradation.

Field trials of bioremediation (Mills et al. 1997) were conducted on a coastal brackish wetland in Texas where two diesel pipelines, one gasoline pipeline and one crude oil pipeline ruptured, spilling ~4,700 tonnes into the tidal and coastal waters of the San Jacinto River. The effect of biostimulation was investigated by evaluating the application of diammonium phosphate alone and diammonium phosphate plus nitrate in replicated plots, providing nitrogen and phosphorus nutrients that otherwise would be limiting. Results showed that the diammonium phosphate treatment significantly enhanced the biodegradation rates of both total GC-MS-resolved saturates and total GC-MS resolved PAHs (Appendix A), and the addition of diammonium phosphate plus nitrate only enhanced the biodegradation of total resolved saturates.

Shin et al. (1999, 2000) investigated the effect of nutrient amendment on the biodegradation of a Louisiana 'sweet' crude oil and oxygen dynamics in a Louisiana salt marsh vegetated by *S. alterniflora.* Significant biodegradation of crude oil in the salt marshes occurred only when the tidal cycle exposed the surface of the marsh to air (Shin et al. 2000). This study again indicated that oxygen availability appeared to control the oil biodegradation process in salt marshes.

Phytoremediation was studied in a freshwater wetland on the St. Lawrence River, QC (Venosa et al. 2002) (**Figures 6.16 and 6.17**), and later on a salt marsh in Nova Scotia (Garcia-Blanco et al. 2007) (**Figure 6.18 and 6.19**). Both studies intentionally released crude oil onto replicated experimental plots. In the freshwater wetland study, the released oil was raked into the top two inches of sediment to simulate penetration of the oil into the anaerobic or anoxic zone. In the salt marsh study, the oil was not raked into the sediment but left on top of the aerobic sediment. Results indicated that addition of nutrients did not result in enhancement of biodegradation of crude oil contaminating the freshwater wetland plots because the oil was in the anaerobic zone where biodegradation is substantially slower. In such a wetland environment, oxygen became limiting at depths within a few millimeters below the surface. However, in the aerobic salt marsh study, the investigators found statistically significant depletion of the hydrocarbons in nutrient-amended plots compared to those not receiving nutrients. This was a clear example of the rhizosphere effect caused by the biostimulation from added nutrients.

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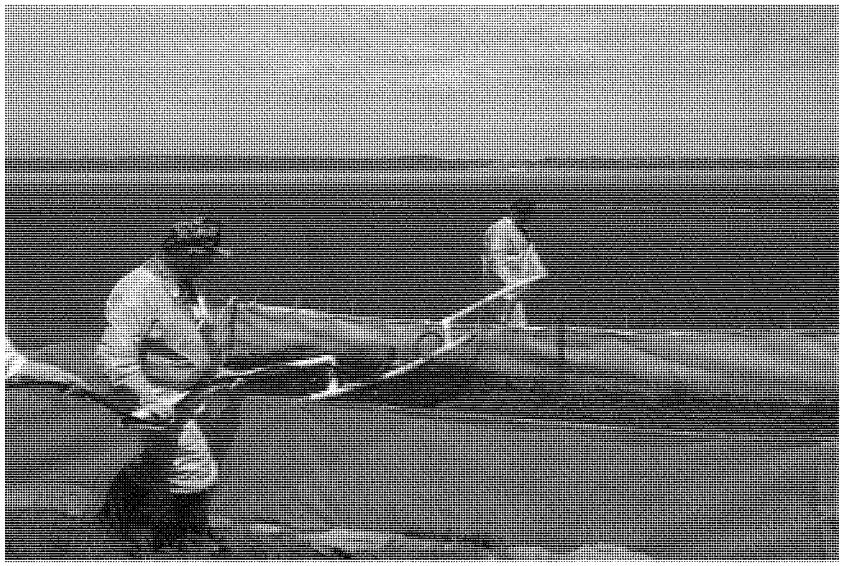

Figure 6.16 Photograph of crude oil being applied to one of the plots in the St. Lawrence field study of freshwater phytoremediation. *Source: Venosa et al. (2002)*

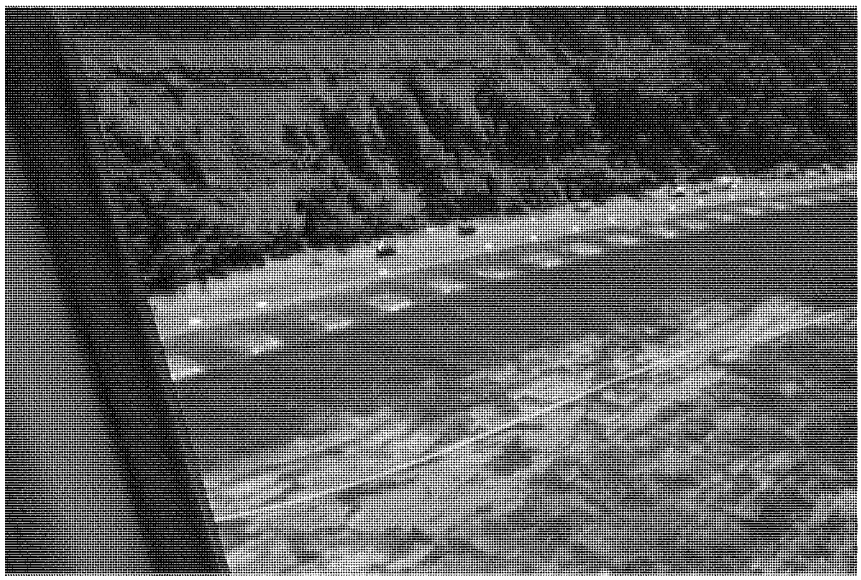

Figure 6.17 Aerial photograph of the plot layout in the St. Lawrence River Field study of freshwater phytoremediation. *Source: Venosa et al. (2002)*

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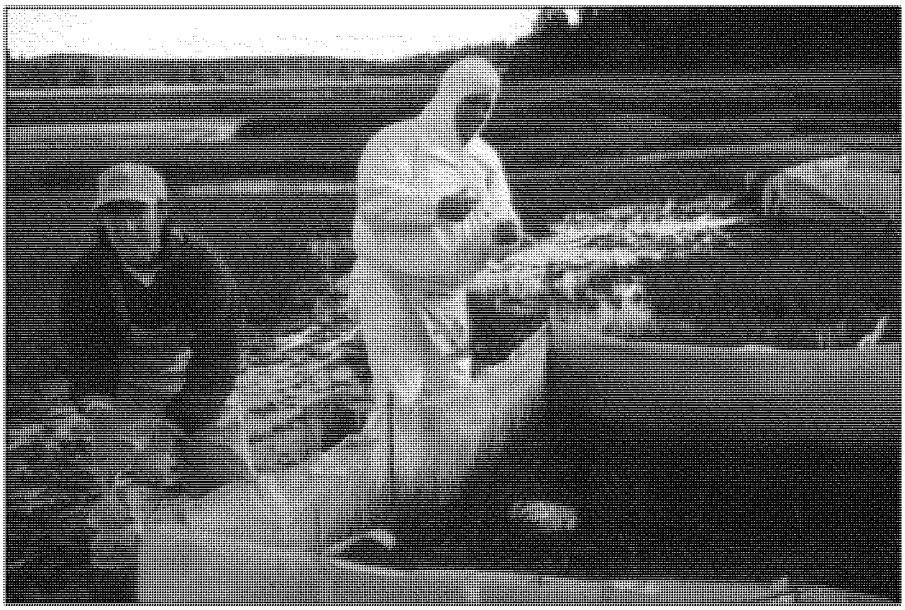

Figure 6.18 Photograph of a team taking samples from a plot in an experimental phytoremediation project conducted in 2001 in Dartmouth, Nova Scotia.

Figure 6.19 Aerial photograph of the experimental plot layout in the salt marsh phytoremediation project in the 2001 Dartmouth, Nova Scotia project.

Another field study on the performance of oil bioremediation in salt marsh ecosystems was carried out in a tropical marine wetland located at Gladstone, Australia (Burns et al. 2000; Duke et al. 2000; Ramsay et al. 2000). This study evaluated the influence of a bioremediation protocol on the degradation rate of a medium range crude oil and a Bunker C oil stranded in salt marshes dominated by *Tecticornia* (a succulent plant) and situated behind a mangrove forest. The bioremediation strategy used in this study involved nutrient addition alone for the salt marsh sites. The addition of the fertilizer to the salt marshes showed a stimulation of the degradation of the crude oil and resulted in about 20% more oil loss compared to the untreated plots. However, the nutrient amendment did not significantly impact the rate of loss of Bunker C oil in the salt marsh plots, likely because of its high proportion of recalcitrant components (Appendix C).

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<u>Recommendation</u>: Most of what we know about oil spill response technologies has been developed through results from studies in the laboratory, mesocosm test systems and case studies (with limited controls and treatment replication). It is the consensus of the scientific community (and the Panel) that controlled field trials are needed to advance our spill response strategies, especially for subsurface blowouts, Arctic oil spills and freshwater shorelines.

6.3.3 Chemical dispersion

Dispersants are chemicals that reduce the oil-water interfacial tension (NRC 2005), thereby decreasing the energy needed for the slick to break up into small droplets and mix into the water column. They are typically applied by spraying from aircraft (**Figure 6.20**) and/or boats. The composition of most commercially available dispersants is proprietary, but in general they consist of one or more nonionic[5] surface active agents (surfactants) dissolved in a solvent carrier (NRC 1989). Some dispersants also include one or more anionic surfactants and other additives. Lower viscosity oils are more amenable to natural and chemical dispersion than higher viscosity oils, such as fuel oils and dilbit.

> The rationale for using dispersants as an oil spill cleanup tool is to reduce wind-driven transport of oil to highly productive coastal waters and sensitive shoreline habitats by breaking the surface oil slick into small droplets to facilitate the transport of the oil into the water column. This would also reduce its exposure to surface water biota (e.g., sea birds). Furthermore, oil in the water column may be diluted to concentrations below the toxicity threshold limits of resident biota. Because microbial attack of oil occurs at the oil-water interface, droplet formation also enhances the biodegradation rates of the residual oil.

Natural dispersion takes place when wave and wind action are sufficiently high (winds > 5 m/s, for example) to overcome the oil-water interfacial tension and break the slick into droplets of various sizes (Figure 2.6). A notable example of such occurred in 1993 when gale-force winds caused the grounding of the tanker *Braer* in Shetland, UK, spilling a low viscosity crude oil into the water. Dispersion occurred naturally, causing minimal shoreline impact. Under less energetic wave action, chemical dispersants enhance natural dispersion, facilitating the breakup of the oil into small droplets that are transported into the water column.

Another form of natural dispersion occurs when microorganisms, especially alkane degraders, produce their own biosurfactants (Bodour et al. 2003). These biosurfactants are similar to the synthetic surfactants in chemical dispersants in their mode of action, which is reducing the interfacial tension between oil and water, solubilizing hydrocarbons and thus facilitating their uptake across the membrane into the cell for metabolic purposes. Bodour et al. (2003) reported that biosurfactant-producing bacteria are widely distributed in both undisturbed and contaminated soils. The majority of hydrocarbon-degrading bacteria reported in the literature belong to the genus *Pseudomonas* (Widada et al. 2002; Bento et al. 2005). A wide variety of metabolic and physiological factors is required for the degradation of different compounds in oil (Frielo et al. 2001). All such properties are not found in one organism but rather in consortia. More research is needed to augment our understanding of the effectiveness of these biosurfactants in enhancing bioremediation of oil-contaminated environments.

Specially formulated products containing surfactants and solvents are sprayed (generally at concentrations of 2-5% by volume of the oil) from aircraft or boats onto the slick. Surfactants are the active (i.e.,

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[5]*Nonionic dispersants (e.g., some alcohols and ethers) do not have an overall electric charge; anionic dispersants are electronegative (e.g., have carboxyl, sulphate or phosphate groups) and cationic dispersants are electropositive (e.g., have amino or ammonium groups).*

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interfacial tension-reducing) ingredients in dispersants. Surfactant compounds are amphiphilic[6] (i.e., have hydrophobic and hydrophilic components within the same molecule), which causes them to accumulate at oil:water interfaces (Porter 1991). Nonionic surfactants are most common in dispersants because they have much lower aqueous solubility than do anionic surfactants (Porter 1991), and they are generally less toxic and less affected by electrolyte concentration than are anionic and cationic surfactants (Porter 1991; Myers 2006).

Figure 6.20 An aircraft releases chemical dispersant over an oil slick in the Gulf of Mexico in 2010. *Source: NOAA /US Coast Guard*

Figure 6.21 is a schematic diagram showing the spraying of a dispersant onto a floating slick and the slick breaking apart into small droplets that get driven into the water column by wave action. The water-loving end of the surfactant is attracted to the water while the oil-loving end is attracted to the oil. The oil droplet is surrounded by the surfactant molecules, forming what is called a micelle or a surfactant-stabilized oil droplet. Eventually, as water mixes with the droplets, their concentration declines and the bacterial communities in the sea are able to break down the oil quickly, leaving no lasting residue that might exert toxicity to the seawater biota.

[6]*Amphiphilic is a term describing a chemical compound possessing both hydrophilic (water-loving, polar) and lipophilic (fat-loving, non-polar) properties.*

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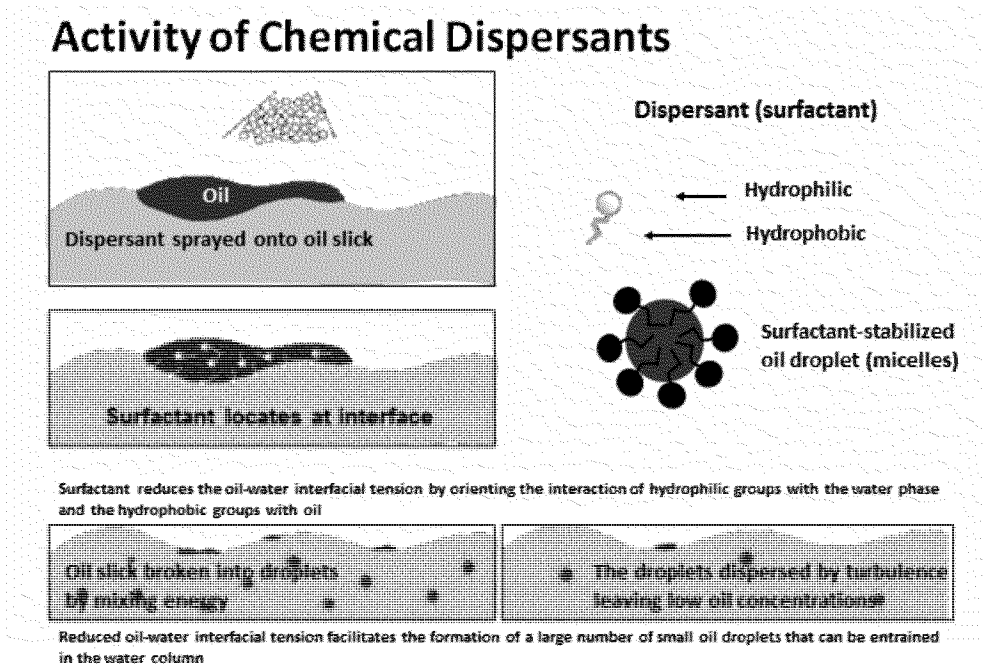

Figure 6.21 Schematic diagram of how oil is dispersed by dispersants.

A typical measure of the relative importance of the hydrophobic and hydrophilic characteristics of nonionic surfactants is the hydrophilic-lipophilic balance (HLB), which ranges from 0 for completely lipophilic or hydrophobic molecules to 20 for completely hydrophilic (charged) molecules. **Figure 6.22** shows the HLB scale ranging from highly oil-soluble to highly water-soluble. Surfactants with low HLB tend to stabilize water-in-oil emulsions, whereas those with high HLB stabilize the more desirable oil-in-water emulsions (NRC 1989; Clayton et al. 1993). Commercial dispersants tend to have overall HLBs in the range of 9 to 11, which is often achieved by combining surfactants having higher and lower HLB. Although the industry consensus suggests that combining surfactants with different HLB improves dispersant effectiveness (NRC 1989; Clayton et al. 1993), others have offered alternative findings (Fingas et al. 1990a).

To date, most chemical dispersants have been formulated for marine use, as the dissolved divalent cations in seawater (particularly calcium and magnesium) are integral to dispersant efficacy (Chapter 3). Development of freshwater-compatible or -specific dispersants has lagged mainly because of regulatory decisions against the use of dispersants near water intakes. A fundamental laboratory study was conducted in 2008 to determine if it is possible to produce a dispersant for use in fresh water (Wrenn et al. 2009). The authors found that dispersants can be designed from scratch (rather than modifying existing commercial dispersants) to drive an oil slick into the freshwater column with the same efficiency as in saltwater as long as the HLB is optimum for that environment.

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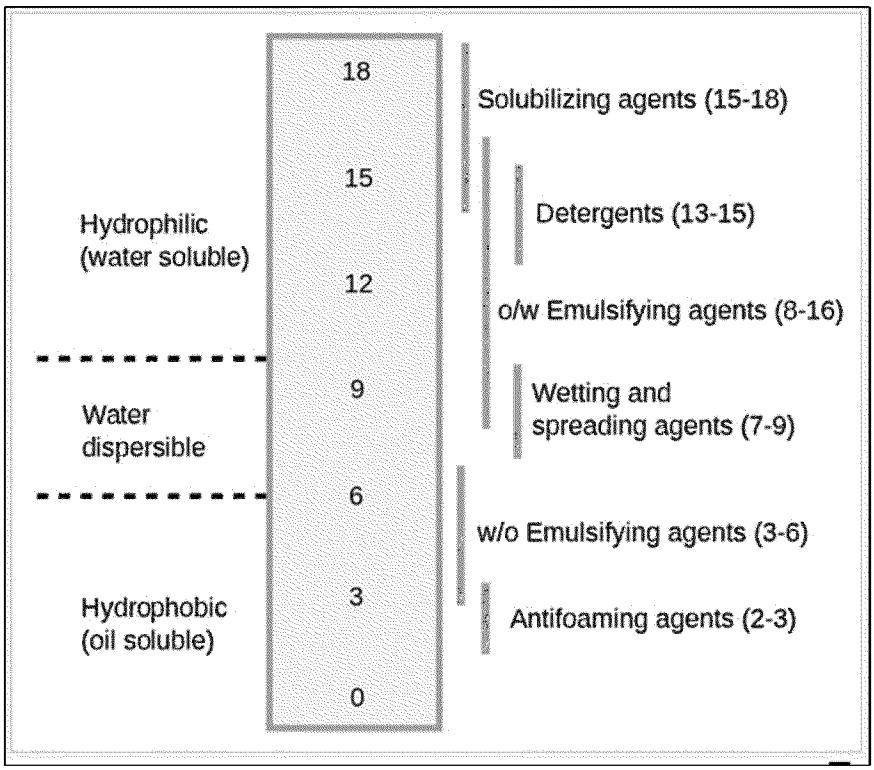

Figure 6.22 Hydrophilic-Lipophilic Balance diagram showing the HLB scale. *(Source: MHK; https://en.wikipedia.org/wiki/Hydrophilic-lipophilic_balance. This figure is licensed under the Creative Commons Attribution-Share Alike 3.0 Unported license)*

> The effect of using dispersants on an oil slick is not simply to dilute the oil. Rather, creating the small oil droplets and driving them into the water column increases the surface-area-to-volume ratio, thereby inducing significantly faster biodegradation by oil-degrading microbial communities within the water.

The purpose of the solvent in a dispersant is to provide the surfactant mixture with an appropriate viscosity, which ensures that it can be pumped through spray nozzles at environmental temperatures. The solvent may be water-miscible[7] (e.g., 2-butoxyethanol) or immiscible (e.g., normal alkanes) (NRC 2005).

Mixing energy driven by wind and wave action is required to achieve effective dispersion. Depending on a site's energy level, tiny droplets of oil (ideally <70 μm in diameter) are mixed in the upper meter of the water column creating a subsurface plume. This plume of dispersed oil droplets rapidly mixes and expands in three dimensions (horizontal spreading and vertical mixing) down to as much as 10 meters below the surface. The rise speed[8] of these droplets is extremely slow and is overcome by the turbulence in the upper depth of the sea, essentially resulting in neutral buoyancy of the droplets. The ideal size of the droplets precludes re-coalescence and re-formation of a surface slick. As a result of this mixing, oil

[7] *Water-miscible solvents mix completely with water to form a single phase, whereas immiscible solvents form two phases.*

[8] *Rise speed is the rate at which submerged droplets will resurface (similar to the concept of terminal velocity for objects falling through air), which depends on droplet size and coalescence; large droplets may take several hours or days to resurface, whereas small droplets may remain submerged for a long time regardless of their nominal density and buoyancy.*

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concentrations decrease rapidly from the initial peak concentrations, e.g., from 10 or 100 ppm down to 1 ppm or less, within hours (Lee et al. 2013).

Microorganisms act on the small dispersed droplets at the oil-water interface (Figure 2.3), so increasing this surface area by reducing the droplet size increases the area for biodegradation to occur. Thus, the ultimate fate of oil is its destruction. The transient effect might be negative to fish and other waterborne animals (Chapter 4), but such an effect may be short-lived due to rapid dilution of the oil to concentrations below toxicity threshold limits, enhanced oil degradation and resilience of the living populations. Dispersants and dispersant applications are rarely 100% effective, however, so some oil will likely remain floating on the water surface.

> In some cases, dispersing oil changes the trajectory of the oil plume from onshore to alongshore, as the transport of dispersed oil is less affected by the wind. Therefore, oil dispersion may help protect sensitive shoreline environments, as wind usually is the dominant environmental factor that carries floating oil to strand ashore.

Dispersant effectiveness is limited by certain physical and chemical variables, such as sea state and oil properties. Sea state has already been discussed briefly above. When wave energy is sufficiently high, re-coalescence of the small dispersed oil droplets is minimized. Salinity is also an important variable. Dispersants in use today have been optimized for use in seawater (salinity of ~35 g/kg). Lower salinities substantially lessen dispersion effectiveness of commercial products unless the dispersant has been manufactured to work in brackish or fresh waters. This is accomplished by manipulating the HLB as discussed previously.

The discussion above would not be complete without mentioning the 2010 DWH blowout in the Gulf of Mexico. In that spill, at least 430,000 to 500,000 tonnes of light oil gushed out of a deepwater well for almost three months before the well was finally repaired and the blowout contained (McNutt et al. 2012; Fingas 2013). Approximately 3,000 tonnes of two chemically similar anionic dispersants (initially Corexit 9527, followed by Corexit 9500, which lacks the miscible solvent 2-butoxyethanol) were injected into the subsurface oil plume as it escaped the well's riser tube and a further 4,000 tonnes were applied to oil at the sea surface (Gray et al. 2011). Significant amounts of data on fluorescence, droplet size distribution, dissolved oxygen, total petroleum hydrocarbons (TPH) and volatile organics analysis (VOA) measurements were collected by the U.S. Coast Guard, NOAA, U.S. EPA, BP, DFO and other organizations deployed in the area and were published daily. Data were obtained from vessels deployed on the surface of the water lowering instruments and samplers to the vicinity of the well and dispersed oil plume, plus autonomous vehicles taking real-time data. These data were important to understanding the hydrodynamic behaviour of the water and the direction of the currents carrying the plume of dispersed oil. Initial findings suggested that the deep subsurface dispersed oil did not experience significant buoyant rise or sinking after release and instead behaved as neutrally buoyant particles or as dissolved materials in the water. All of the measurements indicated that the majority of the deep subsurface dispersed oil was carried predominately to the southwest away from the wellhead by the prevailing currents at depths of 900–1,300 m. Analysis of TPH, VOA and coloured dissolved organic material (CDOM) fluorescence data (Appendix A) indicated the presence of deep subsurface dispersed oil up to 100 km northeast of the well.

Measurements of the dispersed oil plume in the deep sea during the DWH spill at 1,100 to 1,300 m below the surface **(Figure 6.23)** showed that most of the plume consisted of particle sizes ranging from 2.5 to 60 μm in diameter. In the same area, a dissolved oxygen anomaly was observed signifying biodegradation activity. Furthermore, light scattering measurements in the LISST particle size analyzer of water samples collected from that plume displayed the characteristic bimodal distribution typical of chemically- as opposed to physically-dispersed oil (Li et al. 2008, 2009). This suggests, but does not scientifically prove, that the oil in the deep zone was likely chemically dispersed at least part of the time due to the injection of dispersant into the oil exiting from the riser tube (injection of dispersant was not always active). Although

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it is plausible that the extreme turbulence of the oil as it gushed from the well may have caused extensive physical dispersion without the need for chemical dispersant use, review of reported data and information in the literature suggested that it was less likely and that the application of dispersants in the deep sea was successful in dispersing the oil at the source (Venosa et al. 2012). Later, however, Valentine et al. (2014) reported finding oil contamination in the sediment downstream from the well at 1,500 m water depths. The amount of contamination is still being investigated. For a detailed discussion of toxicological issues associated with the use of dispersants in oil spill response, please refer to Chapter 4.

In addition to the dispersant injected at depth into the oil spewing from the wellhead, significant quantities of dispersants were used on the Gulf's surface during the DWH response. Some of the oil that made it to the surface migrated to coastal habitats. The use of dispersants likely increased the bioavailability of the oil (its ability to be taken up or acted upon by organisms) and enhanced the opportunities for biodegradation. However, dispersants also may have increased the potential exposure of sensitive organisms. The fate of the released oil and its many components remains poorly understood. This is an active area of research among oceanographers, engineers and marine chemists. Certainly, the use of dispersants at the sea surface reduced the exposure of the coastline to oil and application of dispersants at the wellhead reduced the amount of oil reaching the surface, but effectiveness of both the surface and the deep sea dispersion was not quantified.

Dispersants have been used successfully as an effective oil spill response technique. For the first time ever, dispersants were used during the DWH blowout to disperse oil as it exited from a deep sea well in the Gulf of Mexico. Although many have concluded that dispersants were successful during the DWH spill, it is still unknown for certain to what extent the oil was dispersed physically due to the high velocity of oil emerging from the well into the water or chemically in response to mixing with applied dispersant.

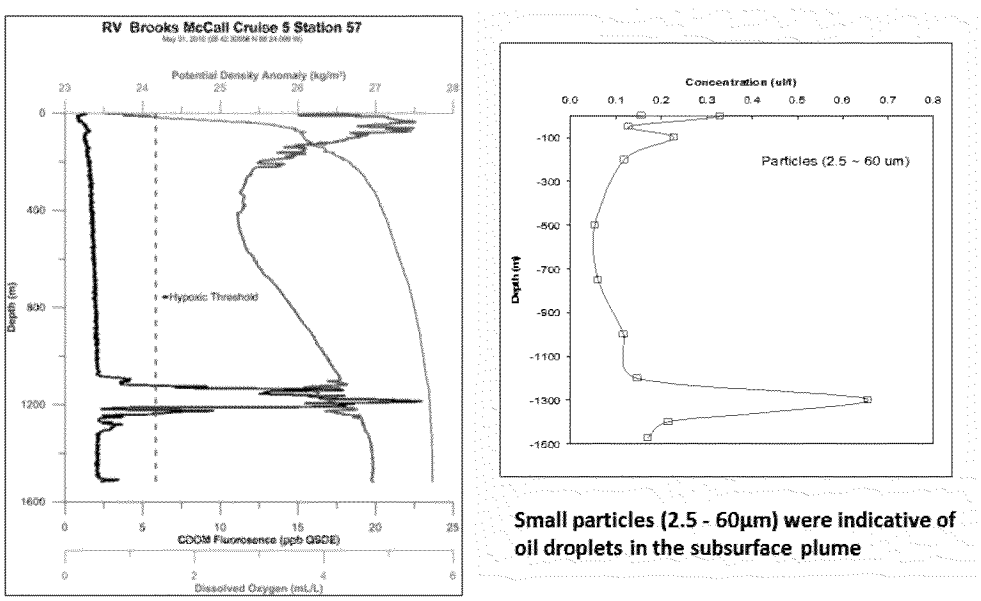

Figure 6.23 Data collected during the Deepwater Horizon blowout showed clearly that at approximately 1200 m below the surface, the dissolved oxygen spiked downward (red curve) suggesting biodegradation while the fluorescence spiked upward signifying the presence of the oil droplets in that same area. *Source: K. Lee, DFO*

<u>Recommendation</u>: The relative importance of turbulence and chemical dispersion needs to be measured in controlled experiments to reduce the uncertainty of the chemical dispersion treatment strategy. Controlled empirical experiments, preferably in the field, need to determine the persistence of dispersed oil in the deep sea and at the sea surface. The ecological and toxicological consequences of dispersants and dispersed oil in wetland, salt marsh, mudflat and deep sea

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environments need to be evaluated at cold and warm temperatures, in part by using mesocosms and wave tanks. Analytical chemical techniques for monitoring dispersed oil or dispersants in the water column at extremely low concentrations need to be improved, especially in fresh water. Research is needed to develop, if practical, effective and non-toxic biosurfactants as alternatives to traditional chemical dispersants currently on the market.

6.3.4 Surface washing by chemical washing agents

Surface washing removes free product stranded on the shoreline without removing sediment. Collected oil is placed in containers and carried off the shoreline. This process is most effective on sand and gravel beaches and riprap[9], but can also be used on other types of terrain except soft mud substrates and mud flats, where the oil would be pushed downward into the sediment causing a worse condition than before. Surface washing is most useful for light to moderate oiling by medium to heavy oils. Removal of surface oil is typically used only along the edges of low, sheltered or vegetated riverbanks and marshes and must be closely monitored.

6.3.5 Solidifiers

Solidifiers are chemical agents applied as powders, granular mixes or gels that react with oil with minimal volume increase to form a cohesive, solidified mass that floats on water (Ghalambor 1997). Use of solidifiers for oil spills on water has been investigated since the early 1970s (Dahl et al. 1997). In some cases, polymeric solidifiers are modified with other inorganic or organic additives, which serve as chemical bonding agents. Due to the need for recovery and disposal of the immobilized oil, solidifiers are generally impractical for wide-scale application to large spills and have been only used on small oil spills on land or restricted waterways (Schulze and Hoffman 1993).

Solidifier booms have also been shown effective in removing sheen from wastewater holding ponds. The U.S. EPA Region 4 authorized solidifiers to be used as an alternative to sorbents or mechanical recovery when removing small or thin sheens from water or small amounts of oil from land (Michel et al. 2008). As reviewed by Fingas (2013a), a solidifying agent manufactured by BP was tested by the Canadian Coast Guard for treating oil-under-ice in the Beaufort Sea and oil in open sea conditions off Newfoundland, with the conclusion that the product's usefulness was equivocal in the first case and not practical in the second case because of the large mass ratio of solidifier:oil required. Hand-spraying the product onto an experimentally oiled Arctic marine shoreline on Baffin Island similarly yielded little benefit (Fingas 2013a). Various other solidifiers have been laboratory-tested by Environment Canada.

Use of solidifiers for oil spill response has some drawbacks and thus has received little attention. For example, their effectiveness depends on the type and composition of the crude oil tested (Fingas et al. 1990b; Dahl et al. 1997). Another disadvantage has been the large amount of material that needs to be applied and recovered. It has been reported that 16 to over 200% by weight (solidifier-to-oil mass ratio) is required to solidify crude oil (Fingas et al. 1995b). Moreover, practical application methods and appropriate tests under various conditions and environments have been lacking (Delaune et al. 1999). In recent reviews of oil spill solidifiers, Fingas (2008, 2013) noted a lack of rigorous scientific assessment of their effectiveness at spill sites. Temperature may play an important role in the solidification of oil mainly because of the variations in viscosity and API gravity of the petroleum product. At low temperatures solidifiers have been found to be less effective (Fingas 2008, 2013a). In contrast, Pelletier and Siron (1999) tested a silicone-based solidifier on a light crude oil and reported that temperature did not appear to have a major impact on effectiveness, although more testing would be required to confirm this finding.

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[9] *Riprap is rock, concrete or other material used to armor shorelines, streambeds, bridge abutments, pilings and other shoreline structures against scour and water or ice erosion.*

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The effectiveness of a solidifier product is dependent upon a number of incident-specific variables, including oil type, oil amount and weather conditions, such as the sea state, sea temperature and wind conditions.

In their review of environmental considerations for non-dispersant chemical countermeasures, Walker et al. (1999) reported that the effectiveness of solidifiers probably decreases for emulsified, weathered, thick and heavy oils because of the difficulty of mixing the product with viscous liquids. They also reported that salinity does not affect the ability of solidifiers to solidify oil on water.

The density and viscosity of the oil reflect the chemical and physical variations of crude oils. Changes in viscosity depend on environmental conditions, particularly temperature. Viscous oils tend to clog the outer layer of the solidifier, thus decreasing further sorption of oil. Fieldhouse and Fingas (2009) reported that product performance was similar at 0 and +15 °C, but at -15 °C longer contact times were required for the samples to reach a state sufficiently solidified to prevent release of oil from the solidified material. The study also reported that low temperature was a limiting factor for generating a solidified product, while at room temperature, the treated material became a cohesive mass. Low temperature was also found to result in longer solidification times or reduced effectiveness due to increased oil viscosity.

Sundaravadivelu et al. (2015) recently conducted an intensive, statistically rigorous laboratory study of oil solidifier performance using a variety of products and oil types at 5 and 22 °C. The removal efficiency and final consistency of the product depended on oil type, temperature, bulk density of the product and application ratio. *Products with lower bulk powder density had high specific surface areas, which resulted in increased oil removal efficiency. For these products, the efficiency was significantly higher at 22 °C than at 5 °C (by 10 to 20%). However, variability was lower as a function of temperature for products with higher bulk density and less removal effectiveness (only 4% increase at the higher temperature). Overall, the best predictor of solidifier effectiveness was found to be its bulk density.*

Recommendation: A laboratory protocol is needed to compare and test effectiveness of commercial solidifier products. Research is needed to determine whether solidifiers are effective or feasible to use on larger oil spills, and whether the product and oil can be recovered after such use.

6.3.6 Herding agents

Herding agents gather thin oil slicks together or move them (at the surface) towards a desired location for collection or ISB, particularly in partial ice cover (Fingas 2013b). Buist et al. (2008) sprayed a herding agent onto the seawater surface surrounding an experimental oil slick, which resulted in the formation of a monolayer of surfactants on the water surface, a process first reported by Garrett and Barger (1972). When the surfactant monolayer reaches the edge of a thin oil slick, it changes the balance of interfacial forces acting on the slick edge and causes the oil to contract into thicker layers. *Since herders do not require a boundary to 'push against', they work even in open water.*

A herding agent is a chemical containing a surfactant and a solvent, similar to a dispersant but in different proportions with different functions.

Surfactants considerably reduce the surface tension of the water surrounding an oil slick (from about 70 mN/m to 20 to 30 mN/m[10]). Although commercialized in the 1970s mostly to concentrate the oil for skimming and recovery, herders were not used offshore because they worked only in very calm

[10] *mN/m (millinewtons per metre) is a unit used to measure surface tension at an interface.*

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conditions. When wind speeds were 2 m/s or greater, booms were needed to contain the slicks, and breaking waves were observed to disrupt the herder layer. *A recent strategy in loose pack ice is to herd freely-drifting oil slicks to a burnable thickness (i.e., greater than about 3 mm). Once a sufficient thickness is attained, the oil may be ignited for ISB. The drift ice acts as an additional containment barrier to aid in reducing the time needed for the development of a slick thick enough to burn. Buist et al. (2008) succeeded in burning oil in drift ice using a non-proprietary hydrocarbon-based cold water herding agent (called USN).*

Based on earlier studies (Buist et al. 2008; 2010a,b), work has continued on herding agents for ISB in Arctic waters (Buist et al. 2011). Recent herding agents were capable of thickening a slick to > 4 mm, suitable for ISB (Buist et al. 2010b), and when tested on experimental oil spills, >90% of the oil was combusted. Based on the results of earlier work (Buist et al. 2008), the researchers concluded that *herding agents work best on relatively fresh crude oil and light distilled product slicks that are still of relatively low viscosity and remain ignitable. Slicks that have gelled or significantly emulsified and viscous residual fuel oil slicks would not be good candidates for herder use*. As with most oil spill response techniques, rapid response improves the chances of success when using herders for ISB.

<u>Recommendation</u>: The utility of herding agents, especially for use in the Arctic, needs to be quantified and properly tested to provide a scientific basis for their application to oil spills.

6.3.7 Debris and detritus removal

All spill incidents involve the production of huge quantities of recovered oil, wastewater, debris, detritus, heavy metals and other solid materials during the spill response that are usually disposed in landfills. Heavy metals are especially important because of their toxicity and persistence in the environment. These cleanup materials include sorbents, debris (seaweed, trash, logs, mooring lines, etc.) and clothing (**Figure 6.24**). Interim storage and permanent disposal options have to be defined. *It should be noted that disposal of contaminated materials in landfills simply moves the pollution from the shoreline to another site.* Accordingly, the work plan should include provisions for reducing the volume of material to be disposed in landfills and should address options, such as recycling, using waste oil in road asphalt and removing excess liquid from the waste oil to decrease volumes. Oiled debris removal is recommended for sand beaches, gravel beaches, sheltered rocky shores, man-made structures and sheltered rubble slopes. It is conditionally recommended for exposed rocky shores, tidal flats, sheltered vegetated low banks and marshes (Region 10 Regional Response Team and the Northwest Area Committee 2015).

In addition, incineration is one of the most efficient methods of disposal for recovered oil, used oil sorbents and debris in a relatively short time and involving not much labour force. The general disadvantage of incineration is high transportation cost of the disposed material to the incinerator facility and regulatory prohibitions in many Canadian jurisdictions. *A recent comprehensive review of all the methods used for treating oily sludge in the petroleum industry was published by Hu et al. (2013), including how to deal with heavy metals and hazardous liquid waste.*

<u>Recommendation:</u> The change of heavy metal concentrations during different oily sludge treatment processes is worthy of investigation in future research, and the integration of available heavy metal control technologies into oily sludge treatment is recommended.

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Figure 6.24 Detritus removal during the Motiti Island oil spill off the coast of New Zealand, 2011. *Source: Maritime New Zealand*

6.4 Factors Affecting Spill Response and Cleanup Effectiveness

6.4.1 Oil types and properties

> The types and properties of spilled crude oil are among the most significant factors influencing the efficiency and effectiveness of oil spill response.

Very light oils (e.g., jet fuels and gasolines) are highly volatile (evaporate in one to two days) and have high concentrations of toxic soluble compounds. Ventilation and/or absorption are usually effective cleanup methods in conjunction with fire and inhalation prevention measures. Light oils (e.g., diesel, No.2 fuel oil, and light crudes, including Bakken crude) are relatively volatile and thus easier to treat effectively than heavier oils. Medium oils (e.g., most conventional crude oils) are generally less volatile than lighter oils, increasing the potential for impacts on waterfowl and wildlife. Heavy oils and diluted bitumen (e.g., Bunker C, weathered dilbit and railbit) are receiving increasing attention due to their resistance to evaporation and biodegradation and their potential long-term damage to the environment, waterfowl and fur-bearing animals. They have fewer components that dissolve in water (Kok 2011; Dejhosseini et al. 2013). Cleanup is extremely difficult for both marine and inland spills. For detailed information about crude oil types and properties, refer to Chapter 2 and Appendix B.

The main physical properties that can affect the behaviour of oil are density, viscosity and flash point (Fingas 2012; Yasin et al. 2013). Oils with high API gravity tend to be free flowing (low viscosity), have a high proportion of volatile compounds and low flash points and pour points, which facilitate their cleanup. However, the properties of oils with high wax or asphaltene content cannot be solely determined by API gravity. Viscosity increases as the oil weathers (Demirbas et al. 2015). Volatility refers to how quickly the oil evaporates into the air. High volatility usually can increase the amount of oil that evaporates into the atmosphere. Flash point is the lowest temperature at which sufficient vapour exists above the spilled oil to yield a flammable mixture (Gülüm and Bilgin 2015). Light crude and oil products have low flash points that are suitable for ISB.

The fate and behaviour of the spilled oil will influence response effectiveness. Abiotic processes at water surface and within the water column include spreading, evaporation, emulsification, oxidation, dispersion,

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dissolution and sedimentation (see details in Chapter 2 and 3). Biotic processes in the water column usually refer to biodegradation and uptake by living microorganisms. Emulsification (high viscosity) and sedimentation can impede the effectiveness of response techniques (Cai et al. 2014b). Weathering processes that influence changes in oil properties can be significantly influenced by ambient environmental conditions. *Thus, comprehensive consideration of the above oil types and properties and their weathering processes, along with environmental factors, are necessary for making sound response decisions and actions.*

6.4.2 Environmental and ecological factors

6.4.2.1 Weather

The feasibility and effectiveness of oil spill response are heavily dependent on the weather conditions prevalent at the oil spill site. Generally, a clear, sunny day without strong wind would be favoured by spill responders, especially in offshore spill sites (Fingas 2012; Doerffer 2013). Such weather conditions would encourage natural attenuation (e.g., evaporation), help the responders deploy equipment and enhance the efficiency of booming, skimming, ISB and dispersant application (Verma et al. 2013).

In hot or warm weather and water conditions, evaporation usually dominates as higher temperature tends to increase the volatility of spilled oil, especially for lighter products. The viscosity of water-oil emulsions decreases with increasing temperatures. Therefore, mechanical cleanup techniques, such as booming and skimming, are much easier to deploy and more effective in warm conditions. Temperate conditions also favor ISB due to the ease of igniting the oil slick. While indigenous microorganisms are adapted to their environment, their rates of activity are generally enhanced at higher temperatures in both the water column and the sediment. *Oil biodegradation rates and extent are generally higher in warm water environments. For example, an increase of 10 °C in temperature can double the metabolic rate of an animal, plant or microorganism (Tyagi et al. 2011; Hassanshahian et al. 2012). The bacterial metabolic rate has been reported to be on the order of 800% higher in the Gulf of Mexico than in Alaska (Mortazavi et al. 2013).*

In contrast, oil spill cleanup is usually complicated by cold and harsh climatic conditions such as those that prevail in northern Canada and the Arctic. Deploying response equipment and personnel is challenging not only because of these harsh conditions but also the lack of access and the long distances between the source of cleanup equipment and the spill. Physical constraints include the fact that during winter months, it is often too cold to work and daylight hours are limited (or non-existent). In addition, *cold weather may cause health and safety issues due to factors such as potential hypothermia among responders, brittle failure of metal components, freezing of sea spray causing vessel instability, etc.* The recovery of oil in extreme cold weather requires a different approach than in warm weather and the likelihood of injury to responders increases significantly (Daling et al. 2010; Buist et al. 2011). Different sets of skills and technologies are needed by responders to ensure effective spill response in harsh environments (Oskins and Bradley 2005).

Cold conditions may also compromise the effectiveness of mechanical containment and other cleaning efforts. For example, mechanical oil recovery, such as through the use of skimmers and pumps, may be compromised because the equipment may freeze up and the increase in oil viscosity reduces recovery efficiency. In the case of ISB, ignition may be difficult and burning may be slow and inefficient. In addition, considerable debate continues over the effectiveness of dispersants on crude oil degradation at low seawater temperatures (Daling et al. 2010). The main concern is that as the temperature decreases, chemical and biological processes slow down and oil viscosity increases, making dispersion of oil more difficult (EPPR 2015). In contrast, reduced oil spreading rates extend the 'window of opportunity' for dispersant use (Nuka Research 2010; Arctic Oil Spill Response Technology Joint Industry Programme

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2013), followed by a reduction in the size of the chemically-dispersed oil droplets, thus enhancing the potential for effective biodegradation in a cold marine environment (Cai et al. 2014a; Greer et al. 2015).

In summary, the effects of weather conditions on oil spill response include:

Warm weather

- Evaporation dominates (higher temperature increases volatility and decreases oil viscosity), resulting in better use of mechanical and ISB responses; and
- Microbial communities grow and metabolize hydrocarbons faster at higher temperatures, speeding up biodegradation.

Cold weather

- Deployment of response equipment and personnel is challenging due to lack of access and long distances to the spill site;
- Daylight hours may be limited or plentiful depending on the latitude of the spill site and season;
- Injury potential to responders is heightened due to the harsh conditions;
- Different skill sets and equipment are needed (chain saws and ice augurs);
- Crude oil biodegradation rate is much lower at low temperatures;
- Oil viscosity increases, which reduces recovery efficiency;
- Cold temperatures may compromise the effectiveness of mechanical equipment;
- ISB ignition may be difficult and burning may be slow at cold temperatures;
- Reduced oil spreading rates extend the opportunity for dispersant use and reduce the size of oil droplets;
- Cold conditions may cause health and safety issues to responders; and
- Mechanical equipment may freeze up.

6.4.2.2 Wave height

Wave height is an important consideration for any recovery method that relies on containment before collection or removal. Waves can impact mechanical spill response by the following factors (Fingas 2012; Andrade et al. 2013):

- Causing boom and skimmer failure or reducing their effectiveness;
- Making vessels difficult to keep on station;
- Creating unsafe conditions for crew to work on deck;
- Making deployment or retrieval of equipment challenging or impossible;
- Causing oil to submerge so that it is no longer available for recovery; and
- Limiting the ability to track and encounter oil.

Oil is often entrained beneath or splashed over booms in short-period wind-waves exceeding 1–1.5 m, although booms can accommodate higher wave heights in a long-period ocean swell (Deng et al. 2013). The issue of boom limitations in high waves also affects the practicality of using ISB in rough sea conditions (Gong et al. 2014). *Steep, wind-driven waves typically impart a more significant detrimental effect on spill response operations than longer period swells. In addition to driving sea state conditions, wind can also impede on-water mechanical response through direct limits to both operating vessels and*

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equipment (Wang et al. 2012; Li et al. 2014a). These conditions can be problematic in harsh northern Canadian and Arctic regions, as well as large inland lakes, such as those in southern regions of Canada and the Great Lakes, which can have severe wave heights during rough weather. However, it should be noted that many Arctic areas experience less severe wind and sea conditions than most open seas and thus may not significantly experience these problems (Dickins 2011). This is because the regional presence of ice dampens wave action and often limits the fetch over which winds might otherwise create larger, fully developed waves.

In summary, extreme wave height and wind can produce unsafe conditions to operate boats, vessels and aircraft. They may also combine with low temperature to further cause wind chill (Gong et al. 2014). For mechanical recovery, these conditions can move booms and skimmers away from oil slicks or make them less or completely non-functional. In regards to ISB, they may impede oil containment and ignition and compound unsafe conditions. Dispersant spray application, use and monitoring are also hampered by these energetic conditions. However, high waves and wind do offer positive effects, such as potential oil movement away from sensitive areas (if in a right direction) and extra mixing energy, promoting natural dispersion and evaporation (Nuka Research 2010).

6.4.2.3 Ice conditions

In an environment with static ice and snow conditions, the first response timelines may be lengthened if oil remains contained, concentrated and trapped for extended periods of time. In dynamic ice conditions, the 'window of opportunity' for spill response operations can be extended if the oil can be tracked until it appears naturally on the surface or until vessels and crews can access it at some distance from the original spill site (EPPR 2015). Tracking oil in or under ice and snow remains a challenge for *in situ* and remote monitoring technologies. *Furthermore, the choice for spill response strategies is limited in harsh conditions, such as those encountered offshore in moving pack ice and rivers in late winter and early spring when they are high and often over-topping river banks and ice floes are moving rapidly or piling into ice jams that cause even more flooding.*

> Existing trajectory models are limited in their capability to model oil fate and behaviour in the presence of a range of sea ice conditions.

Conventional booms and skimmers for mechanical containment and recovery of spilled oil become increasingly ineffective as ice concentrations increase much beyond 10% or more (Lampela and Jolma 2011). For example, ice may tear or move containment booms or clog skimmers or reduce their efficiency and manoeuvrability. The length of boom that can be deployed and maintained under freezing conditions, even for a short period, depends on the severity of the ice conditions. Limited effectiveness is still possible in very open drift ice (10-30% ice cover) and in isolated polynyas[11] within closer pack ice (Wilkman et al. 2014). However, even the presence of small fractions of ice interferes with boom operation and quickly reduces flow to the skimmer head. Any limited containment of oil, which may be possible in open drift ice, requires rugged, high strength booms to withstand contact with the ice. In addition, ice may also create difficulties for deploying fire booms and reduce ISB effectiveness.

The fundamental limitations associated with maintaining and operating booms and skimmers in ice are further complicated in polar areas by the lack of coastal infrastructure and approved on-land storage sites (Pilipenko et al. 2013; Schmidt et al. 2014). The presence of ice may also preclude spill response vessels from getting close to the spill site or to track spilled oil. Due to downtime associated with the clogging of water intakes on vessels, additional ice scouts or ice management vessels may be needed during response operations (Wilkman et al. 2014).

[11] *A polynya is an area of open water within the sea ice pack.*

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To date, dedicated mechanical systems for operations in pack ice have not progressed beyond the prototype stage.

Although ice conditions preclude the effectiveness of booming and skimming, many field observations and experiments have shown that partial ice cover can also act as a barrier preventing oil from spreading as it forms a natural containment system. *With high ice concentrations and a fringe of fast ice, spilled oil may be rapidly encapsulated within the ice leading to thicker oil films and making burning, skimming and dispersant application more effective (Dickins 2011).* Conversely, ice cover can protect the oil from weathering processes, such as evaporation, dispersion and emulsification, and reduce the mixing energy that is needed for effective chemical dispersion (Nuka Research 2010). Although the deployment of fire booms to support ISB operations may be hampered by the presence of ice, it may not be necessary as the ice itself may also act as a boom.

NRC (2014) recently released a comprehensive review of the current state of the science regarding oil spill response and environmental assessment in the Arctic region north of the Bering Strait. A committee of industrial and governmental organizations tasked the NRC to review research activities and recommend strategies to advance research and address information gaps, to identify opportunities and constraints for advancing oil spill research, to describe promising new concepts and technologies, and to assess the types of baseline information needed to monitor the impacts of an oil spill and to develop plans for recovery and restoration. The report reviews oil spill countermeasures applicable to the Arctic, including bioremediation, dispersants, ISB, chemical herders, mechanical containment and recovery, detection and tracking and trajectory monitoring.

The report identified seven research and development needs for improved decision support, including those listed below:

- Improving methods for ISB, dispersant application and use of chemical herders;
- Understanding limitations of mechanical recovery in both open water and ice;
- Investing in under-ice oil detection and response strategies;
- Integrating remote sensing and observational techniques for detecting and tracking ice and oil;
- Determining and verifying biodegradation rates for hydrocarbons in offshore environments;
- Evaluating the toxicity of dispersants and chemically-dispersed oil on key Arctic marine species; and
- Summarizing relevant ongoing and planned research worldwide to achieve synergy and avoid unnecessary duplication.

Recommendation: Initiation of multi-disciplinary research programs is needed to address specific questions related to cold and harsh environments, such as developing sufficient knowledge on the fate, behaviour and impacts of different oil types (in various weathering/treatment states) in snow or ice conditions. Special training of personnel and development of equipment for harsh environments or extreme weather are needed. Better understanding of the ecological effects in cold environments of various oil spill cleanup measures would be beneficial to spill response.

6.4.2.4 Available daylight and visibility

Daylight duration and intensity directly affect oil weathering processes. A series of field spill experiments in the Russian Arctic showed that photooxidation of spilled oil was a more significant process in the 24-hour summer daylight than in more temperate climates (Serova 1992; Ivanov et al. 2005). *However, shorter daylight and lower solar energy in fall and winter, as well as the associated low temperatures, can reduce evaporation and biodegradation activity by microorganisms.*

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Visibility is an important safety factor during mechanical response operations in the field. Low horizontal visibility can impede or prevent on-water mechanical recovery by making it difficult to see and track oil slicks and by making vessel or air support operations unsafe. Low vertical visibility (cloud ceiling) limits aerial observation from rotary or fixed-wing aircraft. Darkness precludes some aspects of mechanical recovery and makes it infeasible to conduct ISB or aerial dispersant spraying (Nuka Research 2010; Jacobs et al. 2015). Darkness also increases risk of accidents and makes it more difficult to deal with emergencies, such as a man overboard (Genwest 2012; Salt et al. 2012).

6.4.2.5 Ecological considerations

Ecological concerns tend to become a major driving force for spill contingency planning and risk management (U.S. Coast Guard 2001; Aurand and Coelho 2006a,b; Webler and Lord 2010). *A drive is underway to improve understanding of the ecological effects of various oil spill cleanup measures to improve operational spill response guidelines. This need is highlighted by recent studies that provided evidence of negative ecological impacts associated with the application of chemical dispersants in the DWH oil spill (Goodbody-Gringley et al. 2013; Kleindienst et al. 2015) and spill mitigation techniques used in the August 10, 2000 Pine River, BC, Pembina pipeline rupture.* The environmental impact in the latter spill included mortality to fish, insects and some wildlife. Moreover, the District of Chetwynd, BC, had to discontinue use of the river, as well as many groundwater wells near the river, as a source of potable water. See more discussion in Chapters 4 and 8.

It will take time to fully understand and quantify the impacts of dispersants in order to create best practice guidelines. *Nevertheless, some mitigation measures that should be considered include: using mathematical tools to optimize the dose and operational processes or developing more potentially eco-friendly dispersants based on biosurfactants*[12] (Zhong and You 2011; Cai et al. 2014b).

Conversely, significant research and practical efforts have been made in bioremediation and a considerable amount of new knowledge has been developed. For example, indigenous microbial species and nutrient levels are crucial factors that are often taken into account during design and application of *in situ* bioremediation due to their impacts on oil fate and behaviours (JRP 2013). Consideration of ecosystem vulnerability is discussed further in Chapters 7 and 8.

6.4.3 Technical and economic factors

A number of technical factors need to be taken into account when arranging and deploying response equipment (**Table 6.2**). The specific operational conditions for each device are usually suggested by manufacturers and/or determined by responders through drilling and field application, although some special or uncommon environmental conditions (e.g., Arctic and extreme weather conditions) are not well addressed mainly due to lack of sufficient knowledge or tests. However, the feasibility of adopting a specific response technique depends on the proximity of equipment needed at the spill site and the time required to transport it there (Li et al. 2014b). In addition, responders' knowledge, experience and skills should also be considered when choosing the technique or equipment (U.S. Coast Guard 2001). Proper training of the response personnel can greatly increase response efficiency and reduce the possibility of having health or safety issues.

[12] *Biosurfactants are surface-active chemicals produced by organisms. They may have more selective activities than manufactured (synthetic) dispersants and are often biodegradable.*

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Table 6.2 Examples of oil spill response equipment and their affecting factors *Sources: Transport Canada (2013) and EPPR (2015)*

Equipment	Affecting Factors
Command equipment	Weather and sea conditions
Dispersant systems	Wave, ice and oil types/weathering degree
Air-inflatable booms	Wave, ice and wind
Skimmers	Wave, ice and oil types/weathering degree
Temporary storage equipment	Capacity
ISB devices	Weather and sea conditions, wind
Vessels	Capacity and compatibility with onboard devices
Pumps and generators	Efficiency and power
Subsea well intervention equipment	Water pressure and sea conditions

For persistent spills, especially caused by heavy oils and diluted bitumen, the cleanup cost may include long-term soil/shoreline remediation and groundwater restoration expenses. The lack of adequate support funds may result in inappropriate or untimely response (Fingas 2011).

It is usually required by law that employers provide a safe and healthy workplace free of recognized hazards, as well as health and safety plans for response crew. Employers must also provide training and required protective equipment to enable the application of different response techniques (Michaels and Howard 2012). Therefore, the conditions at the spill site and the logistic capacity (e.g., in remote locations or under harsh weather conditions) can affect decisions about the most appropriate set of techniques/devices.

A commonly encountered question, and often the most difficult issue to address, is "how clean is clean?", which for the most part is a site-specific question dependent largely on the species at risk. It also invokes question about the intended end use of the spill site after cleanup: whether the site will be restored, remediated, rehabilitated or reclaimed (decision categories more commonly applied to terrestrial or shoreline sites than to open water)[13]. Michel and Benggio (1999) reported how to select the cleanup endpoints to meet the spill-specific requirements. Generally, cleanup endpoints need to be determined in the early stage of response according to general information and knowledge of the spill site and oil type/properties (Sergy and Owens 2007). Ongoing monitoring is necessary to allow for the modification of endpoints or techniques used, if required. Cleanup should proceed as long as it can accelerate the recovery of the spill site and should be terminated if it would slow down the recovery or have no value (Sergy and Owens 2008). *The judgement of clean conditions depends on the availability and quality of baseline data that are sometimes missing or incomplete (refer to Chapter 7).*

6.4.4 Other factors

Response structure and communication may also influence the response effectiveness given that response personnel, such as onsite response crew, offsite decision makers, technical consultants and other stakeholders, need to communicate and follow up with guidelines and regulatory structure for

[13]*Restoration returns the site to the condition pre-dating the spill; remediation removes or reduces (hazardous) wastes from the site to minimize current or future adverse effects; rehabilitation returns the site to a meet a previously determined use plan; and reclamation converts a disturbed site to former uses or to a different, but productive state (Powter 2002).*

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preparedness and response (Transport Canada 2013). The selection of best available response technique should consider local communities and especially aboriginal groups in Canada in case response measures affect traditional use of lands and waters in either the short- or long-term (e.g., fishing, tourism, agriculture and aquaculture) (Colten et al. 2012). The extent to which each response technique is used should be the subject of consultation with local communities. In addition to aboriginal groups, consultations should include user groups (e.g., fishermen), scientists, community members, business, media and non-government organizations and should include the potential for local people to be involved in the cleanup activities (subject to safety and training requirements) (Hoffman and Jennings 2011).

6.5 Research Needs and Recommendations

6.5.1 High Priority Research Needs

6.5.1.1 Cold, harsh environments and controlled field studies

Current technologies, equipment and response personnel are needed to be fully capable of effectively responding to major inland or offshore spills in Northern Canada and Arctic regions. Special skills and equipment are needed both now and to be developed for inland or offshore oil spill responses under cold and harsh conditions. Related to this, most of what we know about oil spill response technologies has been developed through studies in the laboratory, mesocosm test systems and case studies (with limited controls and treatment replication). It is the consensus of the scientific community and the Panel that controlled field trials are needed to advance our spill response strategies, especially for subsurface blowouts, Arctic oil spills and freshwater shorelines.

6.5.1.2 Sediments

Three *in situ* remediation technologies developed over the last 20 years for the treatment of contaminated soils and groundwater have not been tried for remediating bottom sediments contaminated by sunken oil from spills. These remediation strategies are bioventing, air sparging and dissolved air flotation (DAF), all of which can be used to oxygenate anaerobic sediments to stimulate aerobic biodegradation of oil that settled to the bottom. One or all of these techniques may be successful for treating and overcoming the adverse effects of sunken oil in the anaerobic zone of a river or lake sediment. If such advances can be made, it would be a huge help in cleaning up benthic environments that have been contaminated by oil from accidental spills.

6.5.1.3 Dispersants

Although some have concluded that dispersants were successful during the DWH spill, it remains uncertain to what extent the oil was dispersed physically due to the high velocity of oil emerging from the well into the water or chemically in response to mixing with applied dispersant. This question needs to be answered by controlled experiments to reduce the uncertainty of this treatment strategy. Also, questions about the persistence of dispersed oil in both the deep sea and on its surface need to be addressed with controlled empirical experiments. This can be partially ascertained through the use of mesocosms and wave tanks. A further need is to determine if effective and non-toxic microbial biosurfactants, discussed in Chapter 3, can be developed as alternative sources of dispersants.

6.5.2 Medium Priority Research Needs

6.5.2.1 Bioremediation, including phytoremediation and wetlands

Substantial new research on hydrocarbon biodegradation has been conducted in the Gulf of Mexico following the DWH spill in 2010. There is an ongoing need to quantify the persistence of oil of all kinds during and after biodegradation and oil that has been chemically dispersed in the environment following a

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high impact spill to help stimulate innovations in bioremediation approaches. This is especially pertinent to cold temperature and anaerobic environments.

6.5.2.2 Herding Agents and In Situ Burning (ISB)

Recent advances in the study of herding agents (Buist et al. 2008; Buist et al. 2010 a, b; Buist et al. 2011; Lee et al. 2013) have been made that suggest they might be useful in corralling an oil slick for ISB. This may be especially true in Arctic environments. The utility of herding agents for this purpose needs to be quantified and properly tested to confirm this response approach. Such advances could measurably improve the use of ISB in Arctic oil spills. ISB is fast gaining a foothold as an effective response technology for treating a floating oil slick. More quantitative date are needed to determine the emissions that are released to the environment from ISB that may affect downstream ecosystems adversely, such as any incompletely combusted PAHs, toxic particulate matter, etc. It is also important to determine the safety and effectiveness of ISB. More peer-reviewed articles published in the literature are needed in support of ISB as a safe and effective oil response tool.

6.5.3 Low Priority Research Needs

6.5.3.1 Berm Relocation, surf washing, sediment reworking

Berm relocation, surf washing and sediment reworking have been used successfully to accelerate shoreline cleanup in the past. Concerns about this process have been primarily linked to the potential fate and ecological effects of the oil released during sediment relocation. The results of preliminary studies to date suggest that the oil in near-shore waters is diluted to concentrations within regulatory guidelines for ocean disposal for dredged sediments (Lee et al. 2003b). More research is needed to advance the status of this cleanup methodology in terms of fate and effects or released oil.

6.5.3.2 Sorbents, surface washing agents (SWA), solidifiers

Sorbents are widely used as a cleanup tool to combat oil spills in water and on shorelines. They have their limitations, however. The need exists to determine if any advances have been or can be made to improve sorption as an effective response tool, especially in the use of biodegradable natural organic sorbents, such as bagasse, that may possibly not need to be removed for biodegradation to take place. Significant amounts of debris and solid waste are generated when using sorbents for cleanup, so if natural sorbents can be developed that would sorb oil and then remain in place for ultimate cleanup by biodegradation, this would be a significant advancement in oil spill response. There is additional need to determine if any adverse effects on the ecosystem accrue from the use of SWA chemicals in spill response. In regard to solidifiers, recently, new research has been advanced in the understanding of the mechanism of solidifiers in removing oil floating on water. It is unknown if these materials can be made effective on larger oil spills.

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CHAPTER 7: PREVENTION AND RESPONSE DECISION MAKING

Abstract

This chapter emphasizes on the past and current development and practices from two important aspects: preparedness and prevention before an oil spill and response assessment and decision-making after a spill. The review leads to the identification of challenges and future research needs.

Section 7.1 focuses on the research status and current practices in oil spill preparedness and prevention. 1) Prevention - Concerns are growing due to spill risks in the Arctic, some major ongoing or proposed pipeline projects, and the recent spills caused by pipeline raptures and roadside accidents. Continued efforts are identified in understanding the associated impacts and improving prevention strategies. 2) Oil spill monitoring - Advances in remote sensing and airborne imaging have created a promising set of cost-efficient and practical tools in oil spill response. The widely employed shoreline cleanup assessment team (SCAT) approach needs further improvement to integrate with response modeling and decision making. 3) Oil and Sample Analysis - Environmental forensics analysis and biological monitoring have played an ever increasing role in oil spill investigation and response decision making, although future needs are identified such as for the Arctic and dilbit. 4) Baseline data - Challenges and gaps exist, particularly in data acquisition, integrity, sharing/coordination, and uncertainties. 5) Sensitivity and Vulnerability - Incorporation of sensitivity and vulnerability analysis into risk assessment has been increasingly adopted, but quantitative research are limited on integration with modeling and decision making and for the Arctic. 6) Preparedness and contingency planning - The challenge is how to determine what level of preparedness is appropriate for each area of response. There lacks a comprehensive national framework in place for training and exercises that involves all key stakeholders, as well as sufficient capability to manage potential risks to the coast line.

Section 7.2 reviews the research and practice in oil spill early warning, technology selection, mathematical modeling, risk and impact assessment, cleanup process simulation, decision making and information technologies. 1) Early warning - Most existing response systems lack, or are weak in, effective early warning function/ability to identify, diagnose and react promptly to minimize the oil discharge into the environment at an early but critical stage of the emergency. It is challenging to incorporate the selected indicators into early warning and real-time monitoring systems and into spill response decision-making processes. 2) Technology screening and evaluation – Approaches, such as best available technology standards, have been used for response technology selection but difficulties exit in reflecting technical limitations and working conditions varied with environmental and human factors. New/updated guidelines are needed to govern application based on improved knowledge. 3) Spill modeling – Substantial modeling efforts have been made in the past decades but one of the key challenges is how to use/refine modeling tools to more effectively and dynamically couple with response process simulation/control, risk assessment and uncertainty analysis to support response decision-making. 4) Net Environmental Benefit Analysis (NEBA) – With recognition of the value of NEBA, future considerations are identified especially for response actions such as socioeconomic, political and legal factors, operational and logistical constraints, influence of human judgment, ecosystem services, and interactive, collateral and long-term cumulative impacts. The chronic oil leakage from routine operations and discharge of hydrocarbons in wastewater effluent (e.g., produced, ballast, bilge and quench water) have not been given good research attention. 5) Performance evaluation and post-event management - It is critical to assess response performance during and after a spill event to learn lessons for future improvement. Adaptive management especially with integration of risk analysis has been limited in the field. 6) Risk assessment - There is a lack of assessment of future oil spill risks due to increased traffic associated with the opening of routes in the Arctic waters. Concerns are emerging on human health risks including psychosocial impacts of a major spill. 7) Cleanup process simulation and control – Efforts are very limited in simulating, predicting and optimizing cleanup processes (e.g., *in situ* burning, skimming, and dispersion) individually and collectively and their effects on response effectiveness. 8) Response

decision support systems (DSS) – Inadequate decision support is one of the major challenges that limit the efficiency of current response practices. Due to limited attention and investment, e xisting DSS are rare and lack dynamic and interactive support from other modeling tools (risk analysis, spill modeling, NEBA, process simulation, etc.) and field validation. Uncertainty is a major hindrance to improving efficiency and confidence of decision making. These are especially true for the Arctic waters where the window of opportunity is significantly short. 9) Information technologies - Artificial intelligence and geomatics tools have received increased interest and further coupling with modeling and visualization techniques is desired to better address complexity and dynamics of spills and response operations.

The review of oil spill prevention and decision making is summarized in Section 7.3 and then leads to the high-priority research recommendations described in Section 7.4. First, i t is needed to develop modeling methods to simulate and optimize individual and collective cleanup processes (e.g., booming, in situ burning, skimming, dispersion and bioremediation) for supporting response decision making. Secondly, it is urgent to develop and apply oil spill response decision support systems, which can dynamically and interactively integrate monitoring and early warning, spill modeling, vulnerability/risk analysis , response process simulation/control, system optimization, and visualization. Thirdly, research investment is in high demand on trial tests and field validation of new prevention and decision making methods to demonstrate feasibility, increase confidence for implementation, and improve response capabilities. Fourth, research is needed to better quantify modeling uncertainties, evaluate their propagation, and mitigate their impacts on spill response decision making. Fifth, further research and development a re desired on environmental forensics, remote sensing and in -situ measurement, early warning and diagnosis, and biological monitoring to improve spill prevention and decision making. Finally, special research efforts should be given to some emerging concer ns (e.g., diluted bitumen, aging/subsea pipelines, and railcars) and the harsh and Arctic environments for improving effectiveness and confidence of prevention and response strategies and decisions.

7.1 Prevention and Preparedness

All pre -spill strategies emphasize prevention as a key emergency management activity. Prevention focuses on reducing the likelihood of a spill, thus reducing risk. The basic principle of spill prevention is that risk reduction through prevention is much more effective than risk re duction through management of consequences. Prevention is based on understanding the science and technologies associated with oil operations and potential oil spills, and applying that understanding to the specific environment in which any activity is taki ng place, with the goal of managing risk throughout an operation (US GAO 2012). With a high level of knowledge, many potentially high -risk activities can be anticipated and the risk of spills may be reduced. For example, when an oil well or tanker is prope rly designed for the range of anticipated risks, established procedures are followed, equipment is properly inspected and maintained and training is provided. Thus, spill incidences may decline as suggested by ExxonMobil (2013) and the tanker spill statistics shown in Chapter 1. Furthermore, if accidents occur, the causes and outcomes can be analyzed providing new information to be used to update and improve prevention policies and measures.

7.1.1 Prevention

7.1.1.1 Tanker safety and spill prevention

Possessing the world 's longest coastline with a total length of more than 243,000 km, Canada ships 80 million tonnes of oil per year off its east and west coasts (Transport Canada 2015e).*As more shipping and oil exploration are considered for the Arctic, tanker traffic in the Arctic region may increase significantly from the current low levels of less than 15 tankers per year. Therefore, tanker safety and spill prevention are among the top priorities.*

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Safety provisions for oil tankers include navigation safety focused on prev enting tanker accidents, including collisions, groundings and explosions. Navigation safety regulations are set by international and national agencies and include measures such as:

> The *Canada Shipping Act* was established in 2001 respecting shipping and navigation and to amend the *Shipping Conferences Exemption Act, 1987* and other Acts.

- Speed regulations;
- Passage plans that respect safe navigation and the environment;
- Mandatory pilotage zones;
- Appropriate navigation technology; and
- Reporting requirements regarding vessel routing measures (e.g. traffic separation, two -way routes, recommended tracks, precau tionary areas, and areas to be avoided) (Transport Canada 2014).

In Canada, Northern V essel Traffic Service Zones are established where ships carrying over 500 tonnes must check in with officers for inspections of safety and navigation equipment and for examination of potential pollutants. Compulsory pilotage and tug escort services for dif ficult waterways are provided by the four Pilotage Authorities in Canada (WSP 2014). Moreover, joint industry-government guidelines for the control of oil tankers and bulk chemical carriers in ice control zones of Eastern Canada were established in 2012 (Transport Canada 2015b). These guidelines address the special risks of ice damage in certain waters off the east coast during winter and spring and include provisions, such as: having an 'ice advisor' on board when transiting an active Ice Control Zone, travelling at moderate speeds commensurate with visibility and ice conditions, being equipped with searchlights for night time navigation, and obtaining current ice information and a recommended route to follow.

Since 2001, Canadian regulations under the *Canada Shipping Act* have required double hulls for oil tankers built after 1993. The Canadian phase-out date for single-hulled large crude oil tankers was 2010. The phase-in period for double -hulled smaller oil tankers was up to the end of 2014. The International Maritime Organization phase -in period for double - hulled tankers worldwide will be fully i mplemented in 2015.

> The Northern Canada Vessel Traffic Services Zone is established under the *Canada Shipping Act, 2001*. It consists of: a) the shipping safety control zones prescribed by the *Shipping Safety Control Zones Order*; b) the waters of Ungava Bay, Hudson Bay and Kugmallit Bay that are not in a shipping safety control zone; c) the waters of James Bay; d) the waters of the Koksoak River from Ungava Bay to Kuujjuaq; e) the waters of Feuilles Bay from Ungava Bay to Tasiujaq; f) the waters of Chesterfield Inlet that are not within a shipping safety control zone, and the waters of Baker Lake; and g) the waters of the Moose River from James Bay to Moosonee (Government of Canada 2010).

The results of improved tanker safety are reflected in data on the frequency of oil spills due to tanker accidents. Globally, even with increased tanker traffic, there has been a consistent downward trend in large oil spills (**Figures 7. 1**). In and around US waters, the average annual oil spillage (spills greater than 0.14 tonnes or one barrel) by oil tankers and tank barges has decreased by 91% and 67%, respectively, in the decade 1998 -2007 compared to previous decades (**Figure 7.2**). In Canada, the most significant oil spill in eastern coastal waters occurred in 1970, when the tanker *Arrow* spilled over 10,000 tonnes of Bunker C fuel oil off Chedabucto Bay, NS (also refer to Chapter s 1 and 8). Since then, no large spills greater than 700 ton nes have occurred in Canadian coastal waters. The most significant spill in the last 20 years off the British Columbia coast was in 2006 when the ferry the *Queen of the North* sank with 240 tonnes of oil on board, including 225,000 litres of diesel fuel, 15 ,000 litres of light oil, 3,200 litres of hydraulic fluid and 3,200 litres of stern tube oil (BC Ministry of Environment 2015). The annual

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Canadian ship -source spill frequency for crude oil from 2003 -2012, based upon the data from the Canadian Coast Guard Marine Pollution Incident Reporting System, was close to zero for all spill sizes (WSP 2014a,b). The recent WSP reports (2014a,b) quantified the risk of marine spills in Canadian waters and gave an optimistic estimate of future ship-source spills as shown in **Table 7.1**.

Figure 7.1 *Seaborne oil trade and number of tanker spills seven tonnes and over, 1970 to 2013 (Crude and Oil Product; vessels of 60,000 DWT and above; barges excluded. Image from ITOPF (2015))*

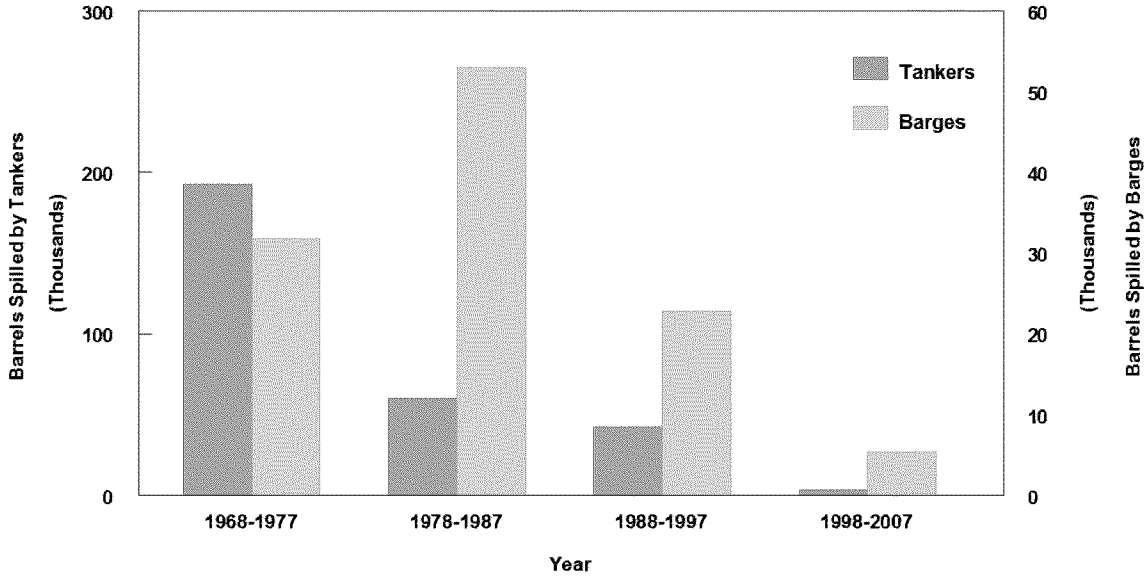

Figure 7.2 *Average annual large oil spills by oil tankers and barges in US Data from API (2009)*

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Table 7.1 *Future Ship-source spills in Canada*[a]

Volume (m^3)	10-100	100-,1000	1,000-10,000	>10,000
Annual Frequency	0.022	0.014	0.019	0.004
Return Period (yrs)	46.4	69.2	51.6	242.3

[a]*Data from WSP (2014a,b)*

7.1.1.2 Pipeline safety and spill prevention

The major existing pipelines carrying crude oil in Canada are (**Figure 7.3**): the Trans Mountain from the Edmonton area to Vancouver and Anacortes, WA; the Keystone pipeline from the Edmonton area through southern Saskatchewan and Manitoba; the Enbridge pipeline from the Edmonton area through southern Saskatchewan and Manitoba before entering North Dakota and then to a network delivering oil to destinations, such as Chicago, Sarnia (via Michigan) and the Gulf of Mexico; the Express pipeline from Hardisty, AB, through southeastern Alberta to Montana and then east/southeast where it joins the network delivering oil to the Gulf of Mexico; and the Enbridge Line 9 running between Sarnia and Montreal. Other feeder pipelines include the Western (Pembina) between Kamloops and Edmonton, the Rainbow pipeline from Norman Wells and pipelines from the Fort McMurray, Cold Lake and Lloydminster oil sands and heavy oil producing areas. With the exception of Enbridge Line 9, all crude oil pipelines in Canada are in the four western provinces.

Due to the strong growth of oil and natural gas production in Canada and the US, pipeline capacity is expected to expand to provide access to new markets. A number of pipeline projects are being proposed to connect the growing supply with replacement markets in eastern Canada and beyond to growing global markets (CAPP 2015*). Although the statistics indicated no significant increase of numbers and volumes of pipeline incidents in Canada since 2008 (* ***Figure*** *7.4), concerns are growing due to major ongoing or proposed projects and the recent frequent spills in North America such as Refugio Oil Spill in California on May 19, 2015 (Panzar et al. 2015), Yellowstone River oil spill in Montana on January 21, 2015 (Yan 2015), Mid-Valley Pipeline Spill in Louisiana on October 13, 2014 (Maykuth 2014), and Hiland Crude Pipeline Spills in North Dakota on March 21, 2014 (MacPherson 2014)*. The National Energy Board (NEB), which regulates pipelines in Canada, recently introduced new pipeline performance measures (NEB 2014b). These measures include training and competency assurance, 'integrity management' (pipeline conditions, equipment inspection, assessment of pipeline hazards, shutdowns for hazard control etc.), integrity inspections, environmental inspections, contractor awareness and damage prevention (e.g., unauthorized activities by other industries or municipalities). The conditions attached to the recent NEB approval of the reversal of Northgate Line 9B reflect these measures (e.g., analysis of threats to integrity of the pipeline, a remaining life analysis, repairs to all features in pipeline sections identified by engineering assessments and a hydrostatic testing program) (NEB 2014a).

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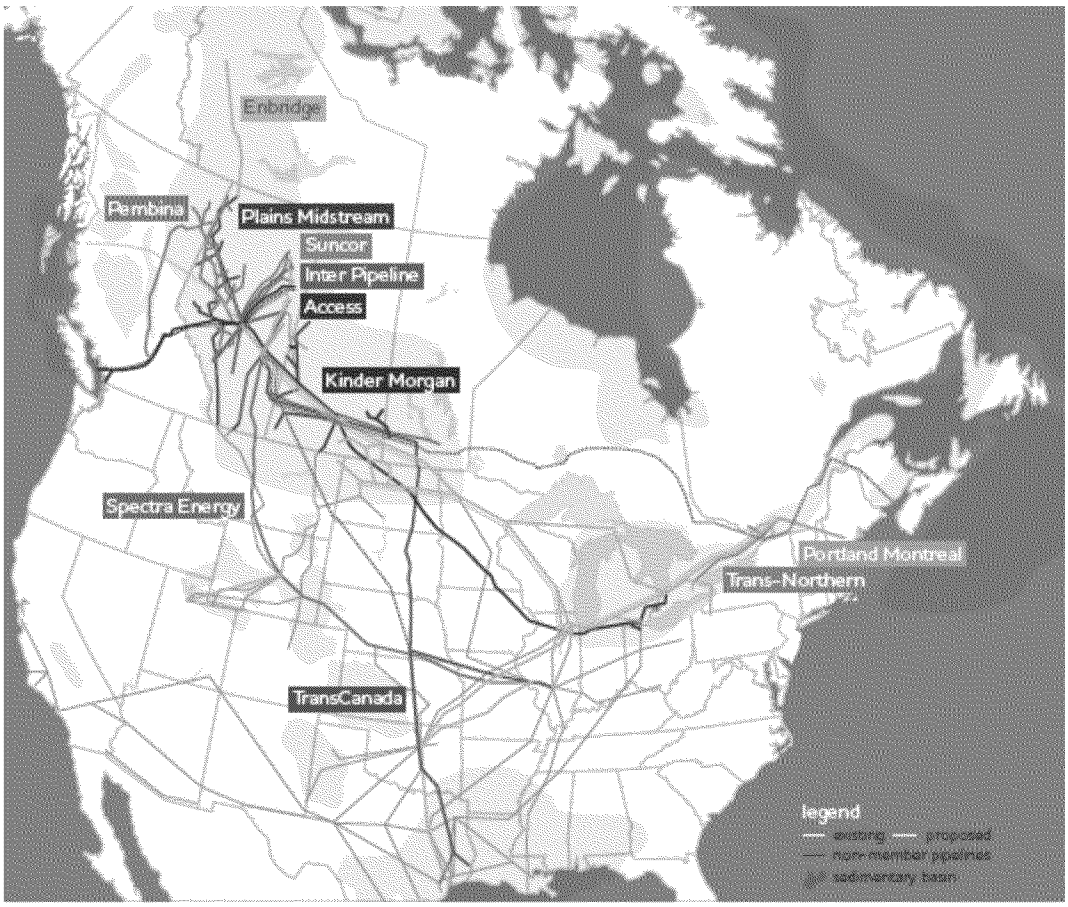

Figure 7.3 *Canadian Energy Pipeline Association (CEPA) liquids pipelines map. Image u sed with permission from CEPA.*

The NEB requires its regulated companies to promote a positive safety culture and expects safety culture programs to apply not only to worker health and safety but to process safety (NEB 2015a). Process safety focuses on preventing catastrophic incidents associated with the use of chemicals and petroleum products. Safety culture is a vital component of spill prevention and in cludes the empowerment of employees to take immediate action (e.g., closing of valves) without waiting for authorization up a chain-of-command.

In 2012, Enbridge filed an application with the NEB to reverse the flow on the segment of Line 9 between North Westover, ON, and Montreal (Line 9B) and that the entire Line 9 capacity from Sarnia, ON, to Montreal be increased with a revision to regulations to allow transportation of heavy crude (NEB 2015b). The NEB had previously approved flow reversal between Sarnia and North Westover in July of 2012. Line 9 originally flowed eastward after its construction in 1975. Flow was reversed in 1998 as oil from areas such as West Africa and the Middle East became more affordable. However, western Canadian crude is now priced lower than foreign oil, leading Enbridge to propose to reverse the flow of Line 9 once again to provide oil to two refineries in Quebec, which represent 20% of Canadian capacity. The NEB approved the reversal on March 6, 2014, subject to 30 conditions which are currently being addressed by Enbridge (NEB 2014a). However, concerns regarding the potential risks and regulatory requirements (e.g., environmental review) associated with using aged facilities have been raised.

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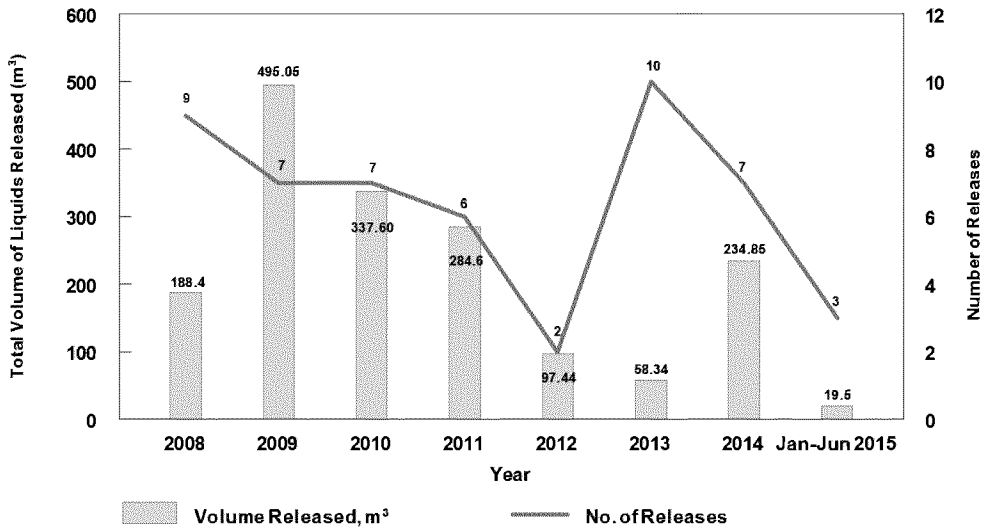

Figure 7.4 *Total volumes of liquids released vs. number of liquid releases in Canada between 2008 to 2015. Image adapted from NEB (2015c)*

Under the *NEB Onshore Pipeline Regulations* (OPR), companies must immediately notify the NEB of any incident that relates to the construction, operation or abandonment of a pipeline. *Generally, spills of crude oil from pipelines in Canada have been infrequent, small (< 1 to 30 m ^{3}) and rarely enter water bodies, although there are variations in volumes and frequencies among provinces and companies (AER 2015). However, some larger incidents have revealed the need for an increased focus on prevention.* Regulatory examination of larger releases in the range of 1,000 -5,000 m^3 in Alberta since 2011 indicated issues, such as: failings in the identification of risks related to corrosion and corrosion repair ; and inadequate leak detection alarm and response processes and lack of training and supervisory oversight (ERCB 2013). For example, a large spill caused by a ruptured year-old pipeline was observed on July 15, 2015, at Nexen Energy's Long Lake oil sands site in Alberta . It released about five million litres of bitumen, sand and produced water and affected an area of about 16,000 m^2 along the pipeline's route (The Canadian Press 2015a). During the writing of this report, Alberta's energy regulator issued an order on August 21 to immediately suspend the company's operations of 95 pipelines in northeastern Alberta due to non-compliance with the *Pipeline Act* surrounding pipeline maintenance and monitoring in its Long Lake oil sands project (The Canadian Press 2015b).

These recent events further raised the attention and importance o f how to prevent and monitor spill incidents in transporting oil by pipelines. Research investment is inadequate to support the development of optimal response strategies and enhancement of response capacities in preventing and remedying spills from aging pipelines for crude oil transport.

Note that Canada-based Enbridge Partners will stop using its No. 5 pipeline under the Straits of Mackinac to transport heavy crude oil under an agreement reached September 3, 2015, with the state of Michigan (Dalbey 2015). This binding legal agreement places a permanent ban on use of the aging pipel ine to transport heavy crude and formally implements the first recommendation of the Michigan Petroleum Pipeline Task Force Report released in July to "prevent the transportation of heavy crude oil through the Straits Pipelines" (Department of Attorney General 2015). The Report gave the recommendation based on the Coast Guard's public acknowledgement that they lack capacity to effectively respond to spills of heavy crude oil in the Great Lakes (Matheny 2014). In addition, Governor Rick Snyder of the State o f Michigan issued an executive order creating the Michigan Pipeline Safety Advisory Board on the same day of signing the agreement.

Information regarding the nature and extent of effects of both smaller or larger pipeline incidents, and the corresponding response actions, is presented in Chapter 6 and later in this chapter.

7.1.1.3 Oil transport by rail: safety and spill prevention

On August 3, 2005, 43 cars derailed just west of Edmonton spilling 700,000 litres of Bunker C fuel oil and 70,000 litres of a wood preservative into the waters of Wabamun Lake, a recreational mecca for many Edmontonians and home to the Paul Band First Nation (Benoit 2007). As admitted by Alberta Premier Ralph Klein at that time, his government was unprepared to deal with the oil spill into Lake Wabamun caused by a train derailment (CBC 2005). A Transportation Safety Board report into the derailment found that CN did have an emergency plan to respond to the spill, but pointed out that it would have been more efficient had there been closer coordination with other agencies (Benoit 2007). The Government of Alberta formed a Commission to investigate the incident and the most significant among the Commission's findings was the absence of a regulatory framework to govern emergency response to major spills of environmentally hazardous goods that fall outside the purview of the *Transportation of Dangerous Goods Act* (Benoit 2007).

Authorities have learned from recent spill events and are gradually improving response management. For example, in response to the derailment and explosion at Lac Mégantic, QC in July 2013, Transport Canada, which regulates the transport of crude oil by rail, took immediate action by establishing a two-person minimum for locomotive crews on trains carrying dangerous goods, and by imposing stricter rules for securing unattended trains (Transport Canada 2015a). Transport Canada has begun a process of enhanced inspections, documentation and follow-up for rail safety, including more frequent inspections at sites where petroleum products are transferred from one mode of transport to another (e.g., from truck to rail). The department has also proposed or introduced fines, mandatory certificates and new safety and transportation of dangerous good regulations and reporting requirements on railways and shippers. Through *Protective Direction 32*, railway operators must provide municipalities and first responders with information on the dangerous goods being transported through their communities. The department has also updated regulations to bring cross-border consistency to the way dangerous goods are identified, providing emergency personnel with a clear understanding of the risks posed by goods being transported.

> After the Lac Mégantic spill, further measures required by Transport Canada have included (Transport Canada 2015a):
> - Removal of the least crash-resistant DOT-111 tank cars from dangerous goods service;
> - New safety standards for DOT-111 tank cars, and requiring those that do not meet the new standards to be phased out by May 1, 2017;
> - Slower trains transporting dangerous goods and introducing other key operating procedures
> - Emergency response plans for even a single tank car carrying crude oil and refined products; and
> - Setting up a task force that meets regularly and brings municipalities, first responders, railways and shippers together to strengthen emergency response capacity across the county.

Railway companies are now expected to have safety management systems, which include safety goals, performance targets, risk assessments, responsibilities and authorities, rules and procedures, and monitoring and evaluation processes. The systems can help companies better comply with federal legislation. Transport Canada monitors compliance through formal safety management system audits and detailed inspections and takes enforcement action in case non-compliance is identified.

According to the Transportation Safety Board of Canada, there were 174 railroad accidents involving dangerous goods in 2014, up from 145 in 2013 and a previous five year average of 131 (TSB 2015). Only one accident in 2014 involved the release of petroleum crude oil. The number and type of rail incidents in **Figure 7.5** illustrates the range of causes, any of which could apply to crude.

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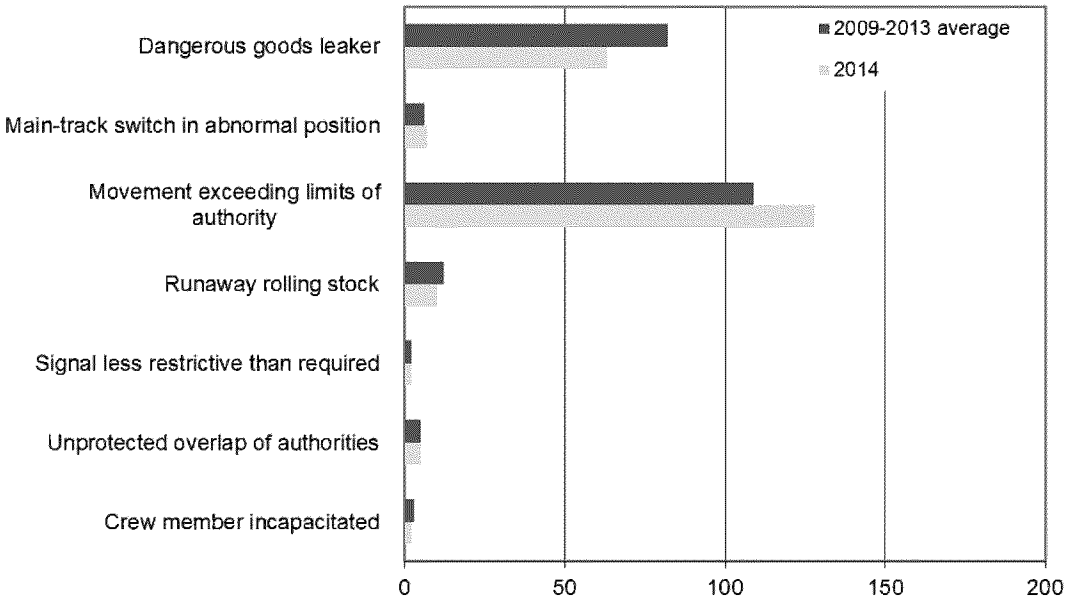

Figure 7.5 *Number of rail incidents by type in 2014. Image from TSB (2015).*

New prevention measures target these causes, as evidenced by Transport Canada's intention to expand mandatory requirements under the *Railway Safety Management System Regulations*. The proposed changes (Transport Canada 2015a) include processes to:

- Encourage employees to report accidents to senior management;
- Analyze data and trends to identify safety concerns;
- Manage organization knowledge so employees can perform their duties more safely;
- Improve work scheduling to prevent employee fatigue; and
- Create annual safety targets and develop tools to achieve them.

<u>Recommendation:</u> Continued efforts are needed to understand the associated impacts of, and improve response management for, spills caused by railroad accidents.

7.1.1.4 Oil transport by truck: safety and spill prevention

The Motor Carrier Division of Transport Canada is primarily responsible for facilitating the reduction of accidents involving large commercial trucks in Canada. Provincial and territorial regulations govern the operation of the commercial truck industry except for the rules and regulations set by the Division in support of the safe operation of extra-provincial motor carriers that carry goods across a provincial or international boundary (Transport Canada 2015c). However, provinces and territories are allowed to regulate extra-provincial truck carriers under the *Motor Vehicle Transport Act* on behalf of the federal government. In addition, the federal *Transport of Dangerous Goods Act* applies to crude transport by truck. However, the provinces are responsible for inspecting vehicles under the Act. Fifteen standards for motor carrier safety are included in the National Safety Code and are supported by provincial regulations. A Safety Management Systems approach is emphasized by the Motor Carrier Division, which is the same approach used to oversee other modes of transport (marine, rail) by Transport Canada. This approach expects the industry to develop a safety culture aimed at identifying and managing risks.

As outlined in Chapter 1, *while the total volume of crude oil transported by truck in Canada is very small relative to transport by pipeline, the number of incidents per volume of crude transported is an order of magnitude greater.* Experience with truck-related oil spills indicates that the largest threats are: contamination of nearby water bodies; effects on drinking water supply or water use for industrial or

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agricultural purposes; fire and explosion and economic impacts caused by property damage; disruption to traffic due to road closures; and disturbance to habitats (Christopherson and Dave 2014). *Therefore, how to decrease the number and severity of crude oil spills involving truck transport is an important issue. Suggested prevention measures could include: improved monitoring and maintenance of key infrastructure at delivery points and fuel loading terminals; identification of high-risk sections of common routes (dense population or highly sensitive environmental areas); and improved truck design and data collection/accessibility for risk management.*

7.1.2 Oil spill monitoring

7.1.2.1 Onsite monitoring

Canada conducts regular aerial surveillance of marine shipping under the National Aerial Surveillance Program. Transport Canada reported that in 2011-2012, surveillance crews observed more than 12,000 vessels and detected 135 pollution occurrences nationally, with an estimated total volume of 1,014 litres of oil (Transport Canada 2015e). There is also an obligation for owners of vessels and operators of oil handling facilities to report marine spills to the Canadian Coast Guard.

During and after spill cleanup operations, it is essential to continuously monitor the sea or site state and weather conditions, as well as the movement and behaviour of the oil. In marine environments, oil spills can be tracked using specially-designed buoys to follow oil movement. Fingas (2012) reviewed oil spill tracking buoys and devices from 1970 until the present and pointed out that the buoys were being beneficially used during exercises and on occasion to simulate oil spills to improve preparedness. Visual monitoring can be undertaken from vehicles, vessels and/or, ideally, from spotter planes or even unmanned aerial vehicles, which can also ensure that the heaviest concentrations of oil are being targeted (Ferraro et al. 2009; Pisano 2011). Lieske et al. (2011) performed a comparative and retrospective analysis of the spatial pattern of oil pollution for the Canadian east coast based on the data obtained from visually unaided aerial surveillance, side-looking airborne radar-assisted aerial surveillance, as well as remote sensing images. The results provided insights into Environment Canada's surveillance programs and offered a useful technique to combine information from different surveillance regimes. Observers should look out for a visible reduction in oil coverage, as well as a change in the appearance of the oil, often seen as a coffee-coloured plume within the water column in good viewing conditions (Chapman et al. 2007; Tan 2011). Submerged flow-through systems using ultraviolet fluorescence spectrometry (UVFS) have also been used to monitor oil concentrations and plumes in water to indicate the effectiveness of dispersion (Lambert 2003; Chatterjee 2015). Although UVFS cannot measure the exact amount of oil, it is able to qualitatively determine whether or not dispersed oil in the water column has increased significantly and therefore whether or not the application of chemical dispersants has been successful (Chapman et al. 2007). The laser fluoresensor is one of the most useful and reliable instruments to detect oil on various backgrounds, including water, soil, weeds, ice and snow. It is the only reliable sensor to date to detect oil in the presence of ice or snow (Brown and Fingas 2003). *However, detection and monitoring of oil spills are always challenging for a number of reasons, such as the substantial land and sea area to cover, low probability and high variability of discharge detection, and limited efficiency, high uncertainty and logistic difficulties due to harsh conditions (e.g., dense vegetation, snow and ice coverage, low visibility, rough waves and darkness).*

In recent years, autonomous underwater vehicles (AUVs) and remotely operated vehicle (ROVs) have been proven especially productive and efficient in spill monitoring. They have emerged as the platform of choice for many surveying tasks, such as bathymetric, near-bottom geophysical and optical surveys. Hundreds of shallow water AUVs (e.g., REMUS) and ROVs (Allen et al. 1997) are operated around the world. Fewer deepwater AUVs and ROVs exist; however, these larger, more expensive systems are routinely used for industrial, military and scientific applications. Notable examples of these systems include the Hugin (Marthiniussen et al. 2004) and Autosub (Griffiths et al. 1999). After the Deepwater

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Horizon (DWH) oil spill, a new 4,500 m depth -rated AUV sentry was developed and applied in tw o separate cruises (WHOI 2010).

7.1.2.2 Remote sensing

Remote sensing technologies have been tested and applied in oil spill resp onses for decades with mixed results (Goodman 1994; Fingas 2011; Salt et al. 2012). Their incorporation into oil spill response requires confidence in the robustness and reliability of the information, based on capturing oil spatial patterns consistent wit h oil spill response processes. Beyond technical challenges, considerations of cost - effectiveness and difficulties in conducting field trials have limited actual field applications to data processing (Leifer et al. 2011).

Recent technological advances in r emote sensing, multi-spectral airborne imaging and echo -sonic tools, along with increased efficiency and lower cost, have created a new set of potentially practical tools for application in oil spill monitoring, preparedness and responses. For example, Yin g et al. (2011) discussed recent research and progress in monitoring sea surfaces covered by oil using microwave remote sensing technology and also proposed a simple and effective filtering technique based on synthetic aperture radar (SAR) for sea oil slic k observation. Leifer et al. (2011) reported a new spectral library approach that uses near infrared imaging spectroscopy data from the airborne visual/infrared imaging spectrometer (AVIRIS) instrument on the NASA ER -2 stratospheric airplane. Extrapolation to the total slick used MODIS (moderate resolution imaging spectroradiometer satellite visual -spectrum) broadband data, which observes sunlight reflection from surface slicks. Ermakov (2013) summarized the results of field studies of the effect of damping of short wind waves due to surface films conducted using radar and optical methods. The results showed that film slicks can be characterized by their specific "spectral contrast sign" and can be discriminated from some look -alikes like wind depression are as. Some radar-based oil spill detection systems (e.g., Rutter's system) have been proven to be effective through extensive field tests and adopted on various platforms, such as floating production storage and offloading vessels, offshore workboats, patrol vessels and specialty cleanup vessels. Etkin et al. (2013) foretold that with more field testing and reductions in costs, some of these technologi es will eventually become more 'tools in the toolbox' for spill response.

<u>Recommendation:</u> Develop improved *in situ* and remote o il leak detection systems to detect spills soon after an incident and post spill monitoring , including autonomous, remotely operated detection devices (e.g., AUVs and ROVs).

7.1.2.3 Shoreline monitoring

When shoreline impact occurs or is likely to occur after a spill, shoreline monitoring and assessment are critical components of the response program, and they provide essential information for setting goals and priorities and making decisions for an effec tive shoreline response (IPIECA 2015). The Shoreline Cleanup Assessment Team (SCAT) approach, which originated from the *Exxon Valdez* oil spill (EVOS) response in 1989, has been widely used in many spills and continuously developed to meet specific spill conditions (Owens and Sergy 2004; Owens and Reimer, 2013). A SCAT survey uses a set of specific and standard terminolog ies to define and assess shoreline oiling conditions, although the process itself is diversified and the assessment activities can be desig ned to match the individual spill conditions (Owens and Sergy 2000 ; IPEICA 2014). The surveys may be applied to spills in both marine coastal and freshwater environments and have become integral components of spill response in Canada, the US and several other countries. *An effective SCAT or shoreline assessment program should be integrated into a shoreline response program and provide critical information to support the contingency planning, response decision-making and implementation/closure processes.*

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The scope and scale of a SCAT survey is governed by a range of parameters that include: the coastal character and configuration, the type and amount of spilled oil; the size of the affected area; and the needs of the response organization. The basic principle s that govern a SCAT sur vey are (Owens and Sergy 2000):

- A systematic assessment of all shorelines in the affected area;
- A division of the coast into geographic units or 'segments';
- A set of standard terms and definitions for documentation; and
- A team of in teragency personnel to represent the various interests of the responsible party, land ownership, land use, land management or governmental responsibility.

The first complete manual for SCAT surveys was developed by Environment Canada (1992), followed by production of a more simplified Field Guide (Owens and Sergy 1994). Other agencies have adopted the approach and produced similar field forms and manuals (e.g., US National Oceanographic and Atmospheric Administration and U.K. Maritime and Coastguard Agency) (US NOAA 1994, 2013; MCA 2007). Environment Canada has developed generic second generation SCAT protocols to standardize the documentation and description of oiled shorelines. The Second Edition of the SCAT manual was developed by Environment Canada in 20 00 particularly with updated standard field forms, which are directly compatible in format and content with the ones used by the NOAA. In 2004, the Arctic Edition of SCAT Manual was published by Environment Canada. It is based on the Second Edition but pro vides new material on the unique shoreline types, characters of the various forms of snow and shore -zone or nearshore ice, cold climatic conditions, and behaviour of oil in arctic regions, as well as on the SCAT activities in these environments (Owens and Sergy 2004).

<u>Recommendation:</u> Future improvement s may focus on how to more effectively and dynamically integrate SCAT into the entire response system covering offshore to shoreline operations, as well as the survey data, into the response modeling and decision-making processes.

7.1.3 Oil and sample analysis

7.1.3.1 Physical analysis

Efficient determination of oil properties can enable effective oil spill response and countermeasures. Some frequently used analytical methods for major oil properties (viscosity, densit y, specific gravity, flash point, pour point and interfacial tension) are summarized in Appendix B.

7.1.3.2 Chemical analysis

Crude oil generally contains hydrocarbons (e.g., paraffins, aromatics, naphthenes and asphaltenes) and heterocompounds (e.g., sulfur, oxygen, nitrogen and metal ions) (Yang 2013a). To be able to unambiguously characterize, identify, categorize and quantify all sources of hydrocarbons entering the environment is important and necessary for oil spill response. Analytical te chniques have been applied extensively in the laboratory to measure and quantify oil constituents belonging to four primary categories: saturated hydrocarbons, aromatics, resins and asphaltenes (Wang et al. 2003) (In -depth discussion can be found in Chapter 2). A summary of the frequently used instrumental techniques is given in Appendix A.

Advances in environmental forensics analysis have played an ever increasing role in oil spill analysis (Wang and Stout 2006). Biological markers or biomarkers (see defin ition in Chapters 2 and 4) are among the most important hydrocarbon compounds in petroleum for chemical fingerprinting (Wa ng et al. 2002, 2003; Yim 2011; Radović et al. 2014). Biomarkers have also been proven useful in identification of petroleum-derived c ontaminants in marine and aquatic environment s. Consequently, biomarker

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fingerprinting techniques have been gradually recognized and increasingly used in characterization, correlation, differentiation and source identification in oil spill investigations (Wang et al. 2012). Biomarkers can be detected in low quantities (μg/L and sub-μg/L levels) in the presence of a wide variety of other types of petroleum hydrocarbons by the use of the gas chromatography/mass spectrometry (GC/MS). For example, Wang et al. (2011) presented integrated forensic oil fingerprinting and data interpretation techniques to characterize the chemical compositions and determine the source of the 2009 Sarnia, ON, oil spill incident. Other examples in the literature confirm the advantages of forensic fingerprinting using these biomarkers (Yim et al. 2012; Yang et al. 2013a, b; Wang et al. 2013, 2014) . Recently, efforts were also reported on identifying suitable biomarkers and applying them for dispersed oil fingerprinting (Song et al. 2014).

Recommendation: Develop and refine environmental forensics technologies for analysis of oil components of interest. Establish standard fingerprinting protocols including new biomarkers to track dispersed oil , diluted bitumens and other petroleum hydrocarbons of interest. Develop protocols to monitor oil in contaminated sediments (on river or sea beds) , particularly in the presence of ice and harsh marine conditions.

7.1.3.3 Biological monitoring

Biological monitoring (or biomonitoring) is the use of a biological community to provide information on the quality or 'health' of an ecosystem (NYSDEC 2015). Basic parameters for biological monitoring include (Reilly and York 2001):

- Mortality of large animals and plants, which are easier to mo nitor (identification of mortality among smaller organisms and plants need special input);
- Sublethal effects, such as bioaccumulation, gene activation, CYP1A induction, behavioural changes or histopathological effects (e.g., presence of disease or injury Chapter 4), which need specialist input;
- Changes in community structure, such as changes in species diversity; and
- Tainting after fish ingest the oil and incorporated into its fatty tissues.

Existing response technologies cause various levels of impacts on the environments. For example, recently interest has increased in the impact of dispersants and dispersed oil particularly focusing on the DWH spill. Judson et al. (2010) discussed the potential for toxicity of the dispersants used in that spill and potential alternatives at the spill site, especially given the limite d toxicity testing information that is available. Using a series of *in vitro* high-throughput assays, eight commercial dispersants were analyzed. White et al. (2012) examined deepwater coral communities three to four months after the well was capped to asse ss the potential impact on offshore ecosystems. The results showed the presence of recently damaged and deceased corals beneath the path of a previously documented plume emanating from the Macondo well providing evidence of impact on the deepwater ecosyste ms. Paul et al. (2013) presented data on general toxicity and mutagenicity of upper water column waters and, to a lesser degree, sediment pore water of the northeastern Gulf of Mexico and west Florida shelf at the time of the spill in 2010 and thereafter. See more discussions in Chapter 4.

Biological monitoring is a proven approach to help collect baseline data before a spill event and evaluate the effectiveness and impact of response and cleanup actions after the event. It can also provide assistance to development of more eco-friendly and cost-effective response technologies.

Recommendation: More research on monitoring parameters /protocols and technologies, as well as result interpretation and integration with decision-making processes, are desired.

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7.1.4 Baseline data

Many environmental and ecological databases and archives hosted by governments have been used in oil spill responses in Canada. Continued efforts have been made to establish and maintain the sources of baseline information at the federal, provincial and local levels. Some examples are as follows:

- National and interagency programs, such as:
  - The GIS Services Group of Canadian Coast Guard maintains databases containing geographic locations of features (e.g. , light stations, aids to navigation, pollution response sites) and events (e.g., search and rescue incidents). The databases are incorporated into a G IS (Dollhopf et al. 2014); and
  - The Canadian Aquatic Biomonitoring Network (CABIN) maintained by Environment Canada to support the standardized collection of biological monitoring information and assess the health of freshwater ecosystems in Canada (Environment Canada 2015a, b).
- Provincial government monitoring programs, such as:
  - GeoBC , which provides a one-stop shop for spatial information for the province, including base maps, specific datasets such as conservation status information for 7,400 plants and animals and fisheries sensitive watersheds, and digital imagery;
  - The BC Burrard Inlet Environmental Action Program, as an interagency partnership (now disbanded unfortunately), compiled baseline data on habitat types, intertidal vegetation, number of bird nests, outfalls, docks, impervious surfaces and numerous other c oastal conditions in the inlet (Chang et al. 2014). These could be instrumental in ranking environmentally sensitive areas to inform a hierarchy of response efforts in the event of an oil spill;
  - The Alberta Environmental Monitoring, Evaluation and Reportin g Agency, a science-based, comprehensive, arm's -length organization responsible for coordinating province -wide environmental monitoring and evaluation with emphasis on the oil sands region; and
  - The Canada-Newfoundland and Labrador Offshore Petroleum Board (CNLOPB), which has continuously produced and compiled the environmental assessment reports conducted by authorized consulting company and made them open for access by the public (Lin et al. 2009; Payne et al. 2012).
- Local conservation authorities, such as the Grand River Conservation Authority in Ontario, which manages water and other natural resources on behalf of 39 municipalities and close to one million residents and operates a network of automatic gauges throughout the watershed to measure river and stream flows, weather conditions and water quality (GRCA 2015).
- Provincial compliance monitoring data for specific permitted facilities which can provide some relevant data near the spill site, such as basic water quality, biological monitoring data and/or facility information.
- Community-based monitoring programs, such as:
  - Streamkeepers training program, a comprehensive education and awareness program supported by Fisheries and Oceans Canada to protect and restore local aquatic habitats (Pacific Streamkeepers Federation 2003); and
  - The Atlantic Coastal Action Program, a unique community -based program initiated by Environment Canada to help Atlantic Canadians restore and sustain local watersheds and adjacent coastal areas (ACAP 2008).

However, challenges and g aps still exist in the baseline data, particularly in acquisition, integrity, sharing and uncertainties. For example, the British Columbia baseline data could not address species' population levels within the Burrard Inlet, which would be helpful for deter mining species most at risk and essential for quantifying impacts of a spill event. Stable legislative and financial mechanisms are lacking in driving long -term data acquisition and management at both federal and provincial levels.

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Access to baseline data and data sharing are still limited among government, industry and academia, which constrains oil spill response research and practices (Lieske et al. 2011; Marty and Potter 2014). *Uncertainties unavoidably exist and come from a variety of sources during data acquisition, processing and manipulation, leading to the need for quantifying the impacts and developing mitigation methods.*

> Comprehensive baseline assessments of natural resources and socioeconomic entities are important for informing environmental and economic impact studies, supporting contingency planning and response decision-making, and facilitating accurate monitoring after a spill has occurred.

In addition, most of those existing sources of baseline data are not well coordinated, accessibility of the data varies and geographic coverage can be spotty, particularly for remote areas. Besides further extending the baseline information, there is a strong need for a thorough examination of available data applicable to establishing a defensible baseline dataset for high-risk aquatic systems adjacent to crude oil shipment routes, whether by marine tanker, pipeline, rail or road. One possible place to start might be a national directory of monitoring information with links to databases, GIS maps and reports.

7.1.5 Sensitivity and vulnerability

7.1.5.1 Sensitivity

Sensitivity is regarded as one of the central concepts in ecosystem protection (Holt et al. 1995; Tyler-Walters and Jackson 1999). It is axiomatic that all ecosystem features have either evolved (in the case of biotic features) or been formed (in the case of abiotic features) within a certain range of environmental conditions.

Sensitivity can be measured using one or more indicators (of species, communities and habitats) that respond to oil spills. These responses are potentially nonlinear and are likely to include interactions among stressors (Adler and Inbar 2007). In this context, sensitivity does not inherently assume the characteristics of fragility or intolerance with which it is often associated. There is no implied judgement that an increased association between the indicator and the stressor reduces a feature's probability of persistence. Nevertheless, as exposure to a chronic perturbation or stress increases, the persistence of that feature is diminished.

> Zacharias and Gregr (2005) defined sensitivity as the degree to which features respond to stresses, such as spills, which are deviations in environmental conditions beyond the expected range.

7.1.5.2 Vulnerability

Vulnerability is the likelihood of exposure to a relevant external stress factor, combined in some way with the exposure (duration, magnitude, rate of change) to that stress. Zacharias and Gregr (2005) provided a quantitative methodology for identifying vulnerable areas based on valued ecological features, defined as biological or physical features, processes or structures deemed by humans to have environmental, social, cultural or economic significance.

Responding to an oil spill is extremely challenging in any environment, especially those regions where extreme weather prevails (Pilipenko et al. 2013). For example, the offshore operating season in the Arctic, and therefore the period when it would be possible to clean up an oil spill, is restricted to four to five months by darkness, heavy ice and extreme cold (Reich et al. 2014). These severe conditions would make it impossible to attempt an oil spill cleanup for half of the time during the operating season and 100% of the time during the winter. In this intervening time, the Arctic environment would be extremely vulnerable to any possible oil spill event. Due to the low temperature and harsh climatic conditions, the

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natural attenuation rate is extremely low in this region (Margesin and Schinner 2001). Given the Arctic region has limited biodiversity, damage to some vital species can lead to widespread changes that cascade through all components of the social-ecological system (Chapin et al. 2004).

7.1.5.3 Vulnerability analysis

> Vulnerability is defined as the probability that a feature will be exposed to a stress (e.g., oil spills) to which it is sensitive (Adger 2006; Metzger et al. 2006).

When responders are making decisions regarding the hierarchy of vulnerable areas to be safeguarded after an oil spill, they need to compare the biological impact and recovery potential, the relative value of the population among other impacted populations and the technical possibilities to help the population. *Incorporation of vulnerability analysis into risk assessment has been increasingly adopted in governmental initiatives, programs and practices*. For example, in 2013, Transport Canada conducted a nationwide risk assessment to determine the risks associated with ship-source spills in Canadian waters south of 60° north latitude and in the Arctic (Transport Canada 2015d). Vulnerabilities have been considered as inputs to risk assessment. On the other hand, *growing efforts have been made to develop more effective methods to classify and evaluate vulnerability to oil spills.* For example, Reich et al. (2014) developed a model of environmental vulnerability to determine the relative vulnerability of broad geographic regions to spilled oil. The model was based on the underlying vulnerability of habitats and representative species present in each region and season assessed. It can be applied to any geographic area of interest and provide a simple, yet comprehensive and expandable methodology for determining baseline environmental vulnerability of large geographic regions. Ihaksi et al. (2011) presented an index-based method that can be used to make decisions concerning which populations of classified natural organisms should be primarily safeguarded from a floating oil slick with oil booms. Olita et al. (2012) proposed a new model for evaluating the hazard of oil slicks contacting shorelines based on the geomorphology of the intertidal area. The hazard index layer can be used as a solid basis to assess the risk and support oil spill response decision-making. Recently, Li et al. (2014a) developed a Monte Carlo simulation-based two-stage adaptive resonance theory mapping model to classify a given site into distinguished zones representing different levels of an offshore oil spill vulnerability index (OSVI). Integrating Monte Carlo simulation with the adaptive resonance theory and mapping approach addresses the uncertainties that widely exist with the environmental conditions. The classification results can also provide the least desired number of zones that can sufficiently represent the levels of offshore OSVI in an area under uncertainty and complexity, saving time and budget in spill monitoring and response (Li et al. 2014c). *However, limited studies focused on the Arctic and sub-Arctic environments are reported in the literature. Vulnerability analysis based on field measurement and integrated with spill modeling and response decision-making are again lacking.*

7.1.6 Preparedness and contingency planning

A management strategy or contingency plan is a set of instructions that outlines the steps that should be taken before, during and after an emergency. It helps minimize potential damage to human health and the environment by ensuring a timely and coordinated response (Chen et al. 2012).

7.1.6.1 Preparedness and response regime

In Canada, the designation of the lead agency for a spill event may be based on legislation, an interagency agreement, a Cabinet decision and/or custom or precedent. There can be more than one lead agency represented under a unified command, as well as the party that is taking responsibility for impact mitigation (e.g., clean up and response management) and generally referred to as either the spiller or polluter. **Table 7.3** shows a list of the lead agencies in Canada for different types of oil spills based on the *National Environmental Emergencies Contingency Plan* (Environment Canada 1999):

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Table 7.3 *Lead agencies for different types of oil spills*

Federal Legislation on Oil Spills	Lead Agency
Deposition of oil into migratory bird habitats (*Migratory Birds Convention Act*)	Environment Canada
Oil pollution from ships (*Canada Shipping Act*)	Department of Fisheries and Oceans; Transport Canada
Cleanup, sampling and analyzing oiled wildlife	Environmental Conservation Service
Oil loading and unloading from ships (*Canada Shipping Act*)	Canadian Coastal Guard
From loading facilities connected to oil production platforms in Canadian Atlantic	National Energy Board

For example, for marine oil spills, Transport Canada is the administrative agency that ensures an appropriate level of preparedness is available for any oil spill incidents occurring within prescribed time standards and operating environments. The Marine Oil Spill Preparedness and Response Regime, established in 1995, is built on a partnership between government and industry (Transport Canada 2013). The regime is designed to ensure that marine shippers pay for spill preparation, response and cleanup. The regime also includes requirements for rigorous contingency planning. The regime is equipped to handle up to 10,000 tonnes of oil spilled in Canadian marine waters. T he Canadian Coast Guard monitors and, where necessary, augments or assumes management of the response when it is in the interest of the public.

Privately-funded certified Response Organizations (RO) have the responsibility to respond to oils spills from ve ssels with which they have arrangements and when contracted to respond. Ship owners are required to have such an arrangement with one or more ROs, depending on the intended destination(s) of the ship and the area covered by each RO. However, there is no legal obligation to enact the arrangement and engage the services of the RO. Alternative arrangements could be made using other resources, if deemed appropriate. Below 60 °N, Canadian waters have been divided into two principal areas: west and east coasts, with an RO established for each. In addition, two further ROs have been established to cover specific parts of the eastern coast region. Each RO has a Response Plan establishing the resources and strategies needed to respond to a range of spill scenarios within its jurisdiction (ITOPF 2013).

Although it is prudent to have some level of preparedness for a worst case spill, the primary focus of the ROs should be preparing for the types of spills that are likely to occur within their area of response. The challenge is how to determine what level of preparedness is appropriate for each area of response. Transport Canada commissioned risk assessments for marine spills in Canadian waters (WSP 2014a, b) as mentioned earlier. These reports estimated the risk of pol lution from marine oil spills to specific sectors of the Canadian coast (Pacific, Atlantic, Estuary/Gulf of St. Lawrence and the Great Lakes/St. Lawrence Seaway System). The results can be used as a valuable base for ROs to determine the level of preparedness and develop their response strategies.

No comprehensive national framework is in place for training and exercises for ship -source oil spill preparedness and response in Canada that involves all key stakeholders. Although Canada's preparedness, at the n ational level, is within an oil spill preparedness and response regime, it still lacks sufficient capability to address and manage the risks existing along the coast line , particularly within the Great Lakes and the St. Lawrence Seaway (Transport Canada 20 13). This lack of capability may be the reason why Canada's current regime has been questioned sometimes in terms of providing the best approach to mitigating the impacts of potential future spills (Marty and Potter 2014; Rise 2014).

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7.1.6.2 Technical preparation

Sound preparedness and contingency planning should rely on knowledge of oil spills and the associated impacts and technologies for response and cleanup. Technical preparation involves not only an examination of spill risks, but also thorough analysis of the nature of the spill events that might occur (source, oil type, spillage rate, location and timing) and the way in which the nature and impacts of spills may change over space and time. For example, Silliman (2014) stated that many environmental factors affect the fate of spilled oil sands products in aquatic environments because bitumen, a large component of oil sands products, has a density greater than fresh water. By analyzing specific factors in areas at risk, responders can better prepare for and expect submergence of oil sands product spills.

If areas identified have low salinity, rough sedimentation, high turbidity, strong sunlight exposure, high temperatures, and strong currents that cause a high risk of submergence, the response teams in these areas should have submerged oil recovery equipment readily available for rapid deployment (Soomere 2012). According to the Joint Review Panel for the Enbridge Northern Gateway Project (JRP 2013a), the advances in technology and modern pipeline design, materials, construction and operating practices would greatly reduce environmental risks from pipeline spills.

A spill of any oil product could have serious effects in any environment. Compared to lighter crude oils, heavier oils and diluted bitumen usually take longer to degrade naturally. The information about oil spill fate, influencing factors and impacts can directly guide the development of contingency plans and the pipelines' design, location and management systems to minimize the potential volume released and keep oil from reaching waterways (Ramseur et al. 2014; Taylor et al. 2014).

Insufficient information could lead to difficulties in making sound contingency planning and response decisions. Many recent publications and conferences, such as the 2015 AMOP Technical Seminar (Environment Canada, 2015c), have revealed knowledge gaps and research needs especially related to the fate and behaviours of diluted bitumen, as well as their implications to response methods under different spill and environmental conditions (especially in cold regions or waters). Also refer to the relevant discussions in Chapter 9.

7.1.6.3 Contingency planning

Oil spill prevention planning requirements are determined by the potential source of the spill, which for transported oil primarily include storage facilities, vessels, pipelines, rail cars and trucks. The designated federal agency must assess the capacity of the responsible party to effectively respond to a spill, which may include providing oversight of response plans, maintaining contingency plans at various levels and personnel training (Ramseur et al. 2014). Transport Canada (2013) proposed an Area Response Planning model that includes a national risk assessment, an identification of area of response, a regional risk assessment, a response requirement, an area response plan, a certification and a continuous improvement plan. *Moving from static national standards to a risk-based model represents an important shift for the Canadian Regime.* The more rigorous planning process inherent in this model may strengthen the links between the public and private elements of the regime and build public confidence in Canada's ability to respond to an oil spill. Provincial governments are also playing a more important role in oil spill contingency planning. For example, the British Columbia Ministry of Environment (2014) recently introduced the preparedness and response guidelines for land-based oil spills, particularly from heavy oil pipelines. The large volume of technical information and stakeholder feedback gathered by the Ministry suggested that world leading practices for oil spill prevention, response and recovery systems to manage and mitigate the risks and costs of heavy oil pipelines are highly desired.

Mathematical models and computer tools have been effectively applied to facilitate contingency planning, although significant amounts of high-quality data, computational resources and maintenance efforts are

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usually required. *Integration with state -of-the-art risk/readiness assessment and geomatic analysis methods has recently been gaining interest and research efforts.* For example, Etkin et al. (2011) recommended a risk-based approach to contingency planning and spill risk management that takes into account the complex factors on both the probability and consequence sides of the risk equation. The approach may include relevant cost-benefit and cost-effectiveness analyses of spill response measures to help plan for the most beneficial strategies to reduce impacts on sensitive natural and socioeconomic resources. Etkin et al. (2013) further suggested that the determination of scope of a contingency plan should be based on the risk of spills within the geographic area that the plan is intended to cover. Lamarche et al. (2013) developed a new cartographic system, based on Google Earth, to support the planning effort in the event of an incident. It allows users to access the system on stand-alone or portable devices, providing a more efficient planning and response tool for spill responders and the responsible party. A variety of geomatic analysis tools and GIS-aided management systems have been increasingly used by Canadian ROs, such as Eastern Canada Response Corporation and Canadian Coast Guard. Taylor et al. (2014) reported that, in 2011, the Regional Association of Oil and Gas Companies - Latin America and the Caribbean developed the *Oil Spill Response Planning and Readiness Assessment Manual* and its assessment tool, the Readiness Evaluation Tool for Oil Spills (RETOS™). The tool provides a general guide to industry and governments to assessing their level of contingency planning and readiness management in relation to pre-established criteria, which are commonly agreed upon by the involved institutions and consider international best management practices. *These criteria are not intended to reflect or add any legal or regulatory requirements. However, their voluntary use by governments and industry can help guide improvements for management of oil spill preparedness and readiness.*

7.1.6.4 Legal requirements and assessment

The responsibility for drawing up contingency plans at a local level, for example, for an individual facility, port or stretch of coastline, and at a larger district or national level, will be dependent on the relevant domestic administrative arrangements. The plan holders should be involved from the outset if plans are to be realistic and practical. Responsibility in the US for ensuring that all plans are compatible usually falls to a national agency, either the US Coast Guard for marine shorelines or the US EPA for inland spills (US Coast Guard 2001). In Canada, to associate industry with government, a legislative framework has been created by an amendment of *the Canada Shipping Act 2001* (Minister of Justice 2001). See Section 7.1.6.1 for details about the lead agencies responsible for various types of oil spills in Canada.

Meanwhile, to guide and enhance response preparedness for transboundary spill events, a number of international contingency plans and legal instruments have also been developed, such as Canada-United States Joint Inland Pollution Contingency Plan, Canada-United States Joint Marine Contingency Plan and the International Maritime Organization (IMO)'s International Convention on Oil Pollution Preparedness, Response and Co-operation (OPRC) (IMO 1990; Environment Canada 1999; Martini and Patruno 2005).

7.1.6.5 Training, drills and education

Training programs should be developed for all response levels and include marine and shoreline response teams and interested parties (US Coast Guard 2001). Regular and realistic exercises will help to ensure that contingency arrangements function properly and that the roles and responsibilities of all parties are thoroughly tested and understood. Equipment should be mobilized and deployed regularly to assess its availability and performance. Such exercises also ensure that contact details and equipment listings are current (Doerffer 2013). *Plans should be periodically reviewed and, if appropriate, amended in the light of lessons learnt from exercises or actual incidents.* All those involved need to be made aware of any changes to the plan (Fingas 2012). Nuka (2010) recommended that field exercises and drills be conducted for training and practice purposes. They can provide valuable information on how well the response equipment perform, the limits to the response systems posed by real-world conditions and the cooperation

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among various personnel and response organizations. *However, exercises are still inherently artificial as the equipment and vessels are pre-assigned and the personnel are pre-notified or even pre-positioned. An unannounced true spill response drill is sometimes necessary to test the operator's response capacity to contact, move and deploy the whole response team.*

Meanwhile, education through social media is being used and should be further encouraged in order to improve the way the community is informed and engaged during oil spills about the situation and how decisions are made. Proper education can help the involved communities improve their attention to preparedness, agility in responding to spills and recovering ability from spills (Merchant et al. 2011). Johnson (2014) suggested a strategy to provide education to local communities in order to help the public understand oil spills and their impacts.

Recommendation: Educational outreach to enhance literacy and promote preparedness and responses should be a key part of the training.

7.2 Response Decision Support

The response to an oil spill usually consists of a series of dynamic, time-sensitive, multifaceted and complex processes subject to various constraints and challenges (Chen et al. 2012). *The success and effectiveness of a response must rely on how efficiently the information (location, oil properties, weather, currents, etc.) and response resources (devices, manpower, money, etc.) can be used and how optimally the decisions and actions can be made.* Even though the policy or regulations focusing on framework or infrastructure are relatively consummate, inadequate decision support may be one of the major challenges that limit the efficiency of spill response. In the past decades, many models have been developed mainly focusing on individual oil spill response activities, including early warning and detection, spill simulation, cost-benefit analysis, risk and impact assessment, cleanup technology selection and cleanup optimization and performance evaluation (Christoph 2013; Lamarche et al. 2013; Paul et al. 2013; Li et al. 2014b,c; Silliman 2014). *However, to date, the integration of the above into an integrated response decision support system is still lacking.* This section will review the current status of research and practice in the area of oil spill response decision making.

7.2.1 Early warning

A reliable, integrated system of early warning and real-time monitoring can significantly improve the effectiveness and efficiency of oil spill emergency response. Similar to medical diagnosis, such a system is able to identify, diagnose and react promptly to minimize the oil discharge into the environment at an early but critical stage of the emergency (Li et al. 2014a).

7.2.1.1 Spill imaging and analysis

Most existing satellite methods of oil spill monitoring tend to be economical, but cannot eliminate the need of systems for early detection and direct monitoring. The integration of *in situ* and remote monitoring hardware with pollution analysis software is of necessity and has been gaining attention. Barenboim et al. (2013) proposed an automated monitoring system for oil spills in waterbodies. It consists of a remote sensing subsystem using fluorescent LIDAR, a network of automatic monitoring stations and an oil pollution identification subsystem based on hydrocarbon contents, alteration of radioactivity and water conductivity. The system provides an efficient tool for early warning and monitoring of oil spills from oil and gas facilities. Rapid distribution of aerial images helps responders at both tactical and strategic levels. In addition, aerial imaging systems that can share information over low cost data links greatly enhance response capabilities by providing real-time information to response teams.

Recent technology advancements enable aerial images to be communicated over low cost data links to stakeholders at different geographic locations within minutes of acquisition. Sweeten et al. (2012)

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incorporated a proof -of-concept test into an emergency re sponse drill that confirmed the ability to transmit images to stakeholders and responders rapidly during oil spill response events.

<u>Recommendation:</u> Comprehensive and in -depth analysis of real -time spill images, along with baseline and historic data, are highly desired to provide more effective and meaningful information for spill response decision-making in a timely and effective manner.

7.2.1.2 Early warning indicators

One efficient way of improving the ability to prevent and reduce major oil spill events and impa cts is to use early warning indicators. **Table 7.4** categorizes the typical early warning indicators and compares their strengths and weaknesses. The use of indicators can be seen as a regulatory requirement and/or a means to avoid unwanted events. Investigation reports of accidents, such as Longford (Hopkins 2000) and Texas City refinery (US Chemical Safety and Hazard Investigation Board 2007), recommended the use of indicators. Øien (2010) compared the different approaches for developing early warning indic ators and suggested that people should be flexible with respect to the choice of methods and preferably use more than one method. One challenge with the establishment of early warning indicators for potential minor events is that these events are not inclu ded in the current quantitative risk analysis used in the offshore petroleum industry (Skogdalen et al. 2011). *Most existing response systems lack or are weak in effective early warning function/ability to identify, diagnose and react promptly to minimize the oil discharge into the environment at an early but critical stage of the emergency. Furthermore, it is challenging to incorporate the selected indicators into the early warning and real -time monitoring systems and into the spill response decision-making processes.*

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Table 7.4 *Summary of early warning indicators and their strengths and weaknesses*

Early warning indicator	Description of approach	Advantages	Disadvantages	References
Safety performance-based indicator	• Describes the safety level within an organization, activity or work unit • Starts with a set of factors that have potential effects on safety • Becomes not only useful for describing safety levels, but also applicable for early warning	• Is favorable when it comes to practicality, simplicity and documentation • Is very relevant as an early warning	• The risk significance and the relative importance between the chosen influencing factors are unknown	Jennings and Schulberg (2009); Skogdalen et al. (2011); Christoph et al. (2013);
Risk-based indicator	• Utilizes risk models as bases, and the development of risk models are part of the method • Regards risk control as the main function of the risk indicators • Becomes preferred with sufficient data • Particularly focuses on organizational risk indicators in the case of early warning	• Provides indicators for major accidents • Easily determines the risk significance • Depends on either accident investigation or occurred events • Indicates potential scenarios without the occurrence of accidents	• It is rather resource intensive, especially for organizational risk indicators, which are desirable since they are most relevant as early warnings	Khan et al. (2002); Pula et al. (2005)
Incident-based indicator	• Depends on detailed analysis of incidents or accidents • Assumes that if the contributing factors are efficient, then neither the incident nor accident have been analyzed nor similar ones have occurred • Mainly focuses on identifying and measuring the factors that contribute to the incident or accident with the use of indicators	• Closely relates to major accidents • Easily communicates with stakeholders based on a factual incident or accident	• Requires very thorough review and documentation • The risk significance and relative importance of the underlying causes are unknown	Jennings and Schulberg (2009); OGP (2011)
Resilience-based indicator	• Questions capability of recognizing, adapting to, and coping with unexpected events by providing specific approaches to manage risk in a proactive manner • Indicates the engineering resilience in organizations and safety management approach with methods, tools and management approaches under complexity	• Focuses on positive signals with failures that may be lack of data • Does not rely on information from occurred events and the indicators are relevant as early warnings	• The risk significance and relative importance of the influencing factors are unknown	Hollnagel and Woods (2006); Øien et al. (2010); Paltrinieri et al. (2012)

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7.2.2 *Response technology screening and evaluation*

7.2.2.1 *Technology standards*

The DWH oil spill revealed a variety of regulatory failures in terms of implementing a regulatory regime, which mandates adequate safety and cleanup technologies in deepwater oil exploration (Bush 2011). To help address the problems, Best Available Technology (BAT) standards are being advocated (Hout et al. 2014; Leifer et al. 2015). In fact, BAT standards have been adopted by some agencies since 1990 and Alaska is among the first. In early 1990, following the EVOS incident in Prince William Sound, Alaska's Department of Environmental Conservation (ADEC) established regulations establishing new oil discharge prevention and contingency planning statutes, which require BAT standards and rigorous technical and economic assessment of BAT (ADEC 1993). However, such an assessment requirement only applies for mechanical oil spill response equipment (Shepherd, 2014).

> Under Alaska state law and ADEC regulations, each blowout prevention contingency plan for marine waters in Alaska must mandate adequate secondary relief well capacity, or similar blow-out prevention and response tools in Cook Inlet. Such plans must include BAT standards "consistent with the applicable" statutory criteria, including: a) whether each technology is the best in use in other similar situations and is available for use by the applicant; b) whether each technology is transferable to the applicant's operations; c) whether there is a reasonable expectation each technology will provide increased spill prevention or other environmental benefits; d) the cost to the applicant of achieving BAT, including consideration of that cost relative to the remaining years of service of the technology in use by the applicant; e) the age and condition of the technology in use by the applicant; f) whether each technology is compatible with existing operations and technologies in use by the applicant; g) the practical feasibility of each technology in terms of engineering and other operational aspects; and h) whether other environmental impacts of each technology, such as air, land, water pollution, and energy requirements, offset any anticipated environmental benefits (ADEC 2015).

In promulgating a BAT regulatory standard, *regulators must identify the best available technology to adequately respond to the particular risks associated with each industrial category. The identification should be based on scientific understanding of the effectiveness and technical/economic constraints especially in cold and harsh environments.* Consequently, regulators must proactively investigate the safety records of oil industry members, the safety regulations and standards, and the industries that are interested in advancing cleanup technology. However, industry usually tends to bypass the standards to reduce the costs (Prendergast and Gschwend 2014). Economic variance can affect the viability of a BAT standard as increased technology requirements often necessitate increased expenditures by industry. Therefore, the oil industry may consider replacing the BAT standard with other strategies to avoid excess costs (Hout et al. 2014). Meanwhile, depending on the location of a spill and weather conditions, response technologies may encounter various constraints to effectiveness (Andrade et al. 2013), especially in Canadian harsh environments where rough weather, high seas, poor visibility, high wind, freezing temperature and ice conditions prevail (Chen et al. 2012). *There have been difficulties in selecting the BAT according to both equipment capacity and working conditions.* Laborde et al. (2015) proposed a framework to develop an oil spill response gap map model based on high resolution historical data at large spatial and temporal scales. The model would show how often environmental factors exceed the operational capacity for *in situ* burning, booming-skimming and dispersant application in a web-based GIS environment.

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7.2.2.2 Screening and performance evaluation

A spill response strategy often starts with the identification of appropriate technologies that accommodate the site -specific environmental conditions and increase the probability of a favourable outcome. Numerous factors must be considered for the selection of oil spill response procedures for use in contingency plans and emergency response operations. These factors include but are not limited to the probability of an oil spill, the possible volume, type and properties of crude oil or refined product that might be spilled, environmental factors influencing the fate and behaviour of the hydrocarbons that could be released, the sensitivity of the most valued ecosystem components (VECs) to oil pollution, the potential impacts from the application of oil spill countermeasures, and the time needed for habita t recovery (Board of Ocean Studies & Marine Board 2014). Leschine et al. (2015) introduced a scenario analysis for oil spill contingency planning and response technology screening for vessels, pipelines, tank farms, berthing docks and many other oil-industry infrastructures. Liao et al. (2012b) proposed an oil spill emergency preparedness system that recommends the BAT based on a comparison with similar cases. *Still one of the key challenges in selecting and developing suitable response strategies and clean up technologies is insufficient knowledge about their technical limitations, influencing factors and ecological impacts, especially related to some emerging concerns , such as diluted bitumen, aging or subsea pipelines, spills to freshwater ecosystems and Arctic conditions. Furthermore, there is a need for new or updated regulations to govern the application of those response technologies based on the improved knowledge.* Further discussion about influencing factors and performance evaluation for response technologies are provided in Chapter 6 and later in this chapter.

7.2.3 Spill physical and numerical simulation

Besides the gaps in modeling oil spills as discussed in Chapter 5, *challenges also exist in how to use modeling tools to more effectively and dynamicall y support response decision-making* (Chen et al. 2012; Li et al. 2012a, 2014b; Li 2014). *The linkage between modeling of oil slicks (weathering and trajectory) and cleanup processes (e.g., in-situ burning and skimming) is lacking, as well as their dynamic c oupling with ecological/ environmental risk assessment to provide decision -makers more accurate a nd comprehensive information. Furthermore, harsh environments prevailing in northern regions and the Arctic can pose large amounts of uncertainty and difficult ies in using the traditional physical and numerical simulation models and their integration with the corresponding response simulation and decision support systems.*

7.2.4 Net Environmental Benefit Analysis (NEBA)

Net Environmental Benefit Analysis (NEBA) has become widely used to assess oil spill countermeasures, both active (e.g., *in situ* burning and mechanical recovery) and passive (e.g., monitoring of natural attenuation processes). NEBA can help determine whether or not additional environmental damage could be caused by specific response actions (Fritt-Rasmussen et al. 2013). NEBA can provide decision-makers with a reliable strategy for deciding what response options are appropriate at a specific spill location based on the analysis of environmental tradeoffs that may occur from the use of the various responses (Coelho et al. 2013; Coolbaugh et al. 2014).

The generic NEBA framework is outlined in IPIECA (2000). A typical NEBA process involves: a) review of previous spills and experimental results that are relevant to the area an d to possible countermeasures; b) assessment of likely environmental outcomes if the proposed countermeasures are implemented compared to outcomes if the area is left to recover naturally; and c) comparison and weighing of advantages and disadvantages of various potential responses with those of natural cleanup. The NEBA process can be used to establish the most important resources at risk before or in an oil spill based, for example, on their status as protected species, ecosystem service or relevance, economic value or human use (Le Floch et al. 2014; Prince 2015).

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The determination of priorities will depend on the preparation of a list of local environments at risk, the results of predicti ve oil weathering and trajectory models, and *in situ* or remote monitoring data, etc. NEBA account s for the nature of the spilled oil and changes it may undergo during weathering and spreading, which may influence the level of environmental, biological and socioeconomic effects (Daling et al. 2014). However, *NEBA can only process environmental variables and does not account for other considerations in the strategy -forming process. A strategy that is environmentally or ecologically ideal but does not recogniz e political or legal factors or operational feasibility ultimately may not be implemented (Coelho et al. 2013). This should be better addressed in the NEBA.*

NEBA tends to be an integral part of contingency plans because post -spill decisions are best and mo st quickly made in light of pre -spill analyses, consultations and agreements involving all of the appropriate organizations and parties (Daling et al. 2014). Use of NEBA in contingency planning offers several advantages, including extended timeframes for a nalysis, consideration of spill scenarios covering a wide range of environmental factors (e.g., seasonal changes in species diversity and ice cover), time for identification and collection of scientific data and stakeholder involvement (Cox 2014). NEBA can also facilitate the identification of potential conflict areas and possible solutions through consultation before any spill occurs. *Other important considerations during NEBA include the logistical constraints that are likely to be encountered during oil spill response operations, which will influence the efficacy of current countermeasure strategies* (Bejarano and Mearns 2015).

Cumulative impacts refer to the effects of an action that are added to or interact with other effects in a particular place (e.g., an oil spill site) and within a particular time (US EPA 1999). The combination of these effects and any resulting environmental changes caused by response actions should be considered in a cumulative impact analysis, which has been incorporated into envir onmental impact assessment processes. *When introducing cumulative impact analysis to NEBA for oil spill response, difficulties may be encountered when identifying/quantifying cumulative impacts of response actions (e.g., application of dispersants), especially for long time periods (Paul et al. 2013; Davies and Hope 2015).*

Regardless of what and how response method(s) are applied, there will be some level of environmental impact. Many studies have conclusively shown that the application of aggressive cleanu p operations may delay the rates of habitat recovery by causing additional damage beyond the oil spill itself (Judson et al. 2010; Kirby and Law 2010). For example, in the aftermath of the EVOS incident, excavation and washing of rocks to remove surface an d subsurface oil were shown not to offer a net environmental benefit because the procedures altered shore structure and delayed biological recovery (US NOAA 1990). The recent DWH spill involved around 47,000 people and 7,000 vessels during *in situ* burning, mechanical recovery and application of dispersants, yet long -term side effects of those activities, especially some caused by the use of dispersants, have been reported (Graham et al. 2011). *In fact, a response strategy that provides protection for one en vironmental resource (e.g., chemical dispersion of oil slicks to protect seabirds) may increase risks to another* (e.g., toxicity of dispersed oil to fishes in the water) (Graham et al. 2011; Payne et al. 2012; Paul et al. 2013). *Decision-makers need to sel ect the optimal response strategy based on the protection of prioritized environmental resources and the countermeasures that offer them the greatest protection, which always lead to some level of trade -off.* The value (effectiveness and impact) of a partic ular method will depend on the situation, including the type and properties of oil spilled, weather conditions, organisms and ecosystems that are impacted and av ailability of response support.

A NEBA process can help identify the relevant factors affecting the effectiveness of options and select the best response strategy. However, *prioritizing response options in NEBA usually involves human inference and judgement regarding making the trade -off (Prince 2015).* Qualitative evaluation approaches, including questionnaire surveys, are usually used and unavoidably introduce uncertainties into the results due to incomplete information and subjective judgement based on personal knowledge, experien ce and opinion (Reynolds 2014).

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<u>Recommendation:</u> More unbiased measures and better understanding of the influence of human judgment are required in NEBA and subsequent decision-making processes. Furthermore, socioeconomic factors play an important part in response decisions and should be reflected in the NEBA process in terms of impact on a region's environment, its economy and the wellbeing of its people.

Many oil spill countermeasure technologies (e.g., chemical agents including dispersants and facilitation of oil mineral aggregate formations) may not be fully understood in terms of their environmental and ecological impacts, and their effectiveness in harsh environments such as ice conditions (Payne et al. 2012; Coelho et al. 2013). Until sufficient information is available on the population dynamics of VECs, the fate and environmental effects of oil and the efficacy of various response technologies, the effectiveness of NEBAs will vary and outcomes will be uncertain (Fitzpatrick et al. 2012; Coolbaugh et al. 2014). *However, application of NEBA is still necessary to assess some newly emerged concerns (e.g., dilbit), countermeasures (e.g., use of biosurfactants as dispersants) and special environmental conditions (e.g., ice coverage, permafrost and peat lands).*

In terms of damage assessment, an ecosystem services approach has been suggested (Menzie et al. 2012; NRC 2013). It can account for an event's impact on all aspects of human wellbeing and can help remediate the damage to natural resources caused by oil spills. Chapters 4 and 8 present the definition and detailed discussion about ecosystem services modeling.

Although NEBA is much desired in the contingency planning stage before a spill, it has also been used to choose the intensity of the response countermeasure, as well as the decision on when to conclude the response (DeMicco et al. 2011). Decisions are typically made based on potential environmental and socioeconomic impacts of the spill, the potential impacts of the cleanup operation and the possibility and ability of natural or human-enhanced recovery of the impacted environment. The 2012 report of the American Petroleum Institute (API), *Spill Response in the Arctic Offshore* (Potter et al. 2012), identified where a range of different cleanup intensities and conditions might be appropriate. It was noted that the most intense efforts to remove oil are appropriate in areas with high levels of human activity or environmental sensitivity, the risk from oil is high and the impact from intense cleanup measures is low. By contrast, monitoring may be sufficient in situations where the spilled oil poses little risk, the potential for natural oil degradation and natural recovery are high and the risks posed by cleanup efforts are high. Another example from the API report showed that, where areas adjacent to the spill site are highly sensitive to oil and the oiled area is only moderately sensitive to cleaning, the appropriate cleanup efforts may include removing most of the visible oil but allowing traces of oil to remain.

7.2.5 Performance evaluation and post-event management

7.2.5.1 Performance evaluation and impact assessment

The oil types, applied response methods, performance and environmental impacts of some major oil spills in Canada and the US are summarized in **Table 7.5**. The summary presents significant variations of performance and impacts of similar response options in different spill events. Post-event comparable studies are lacking to help develop more comprehensive quantitative methods for response performance evaluation in a holistic manner. Consequently lessons from the past spill responses should be better analyzed and reflected in planning and decision-making processes for future spill responses. Also, *effective environmental monitoring and impact assessment practices should be in place, using the most appropriate scientific techniques, to enable the response, conservation and scientific communities to properly assess the real significance of spills and subsequent response activities and to learn lessons for future events. The requirements for improved on-site and post-incident monitoring and assessment continue to increase. Although most risks can usually be estimated during or right after spill response, the variability of the situation may require some degree of on-site or post-event assessment to*

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*continuously assess the risk of oil spill and response activities (*Leong 2012). Such variability can include, but not be limited to, weather changes and subsequent work activities, etc. On -site and post -event assessment usually includes visit s to key risk areas, direct observation, data collection and analysis, questionnaire survey and interview and long -term site monitoring (Dix et al. 2011; Shupe et al. 2014). Bostrom et al. (2015) applied a mental models survey approach to assess public thi nking about oil spills and oil spill response. The decision model considered controlled burning, public health and seafood safety. The results illustrated opportunities to reframe discussions of oil spill response in terms of trade - offs between response op tions, and new possibilities for assessing public opin ions and beliefs during events.

7.2.5.2 Adaptive management

Adaptive management, as a systematic process, has been widely used to continually improve management policies and practices by adjusting subsequent actions based on both past lessons learned and advanced knowledge from new scientific and socioeconomic info rmation on the systems being affected (Rost et al. 2014; Kulapina et al. 2015). The key steps usually include learning and reducing uncertainties, using learned knowledge to change policy and practice, focusing on improving management effectiveness, applying trial management and being formal and systematic (Cluzel et al. 2012; Hausberger et al. 2012). *The application of adaptive management, especially with integration of risk analysis, has been limited in the oil spill response field.* Peltier et al. (2014) developed a new approach that used multiple indicators, including oil slick drifting, to provide monitoring information on dolphin populations and to setup adaptive management strategies. *Adaptive management should be further encouraged to continually improve management policies and practices by adjusting subsequent actions based on learning from past events and new knowledge development.*

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Table 7.5 *Major oil spills in North America and their response performance*

Oil Spill Case	Oil Type	Date	Amount (US gallon)	Responses and Performance	Key Impacts	References
Nova Scotia Uniacke G-72 Oil Spill, Nova Scotia, Canada	Mud, gas and condensate	February 22, 1984	126,000	• Well crews evacuated • No immediate response • Well recapped 10 days later	• Strong hydrocarbon odour was present as far as 10 km away • Oiled seabirds and seals were observed • No evidence of condensate on the shoreline	Gill et al. (1985)
Exxon Valdez Oil Spill, Alaska, USA	Crude oil	March 24, 1989	11-38 million	• Controlled burning (with ship and boom) - 113,400 L of oil reduced to 1,134 L of removable residue • Mechanical cleanup (with boom and skimmer) - only 10% oil recovered • Skimmer: low efficiency (thick oil, heavy kelp, plenty of permanent container, bad weather, etc.) • Chemical dispersants ineffective due to lack of wave energy • Shoreline washing • Bioremediation of shorelines	• Immediate effects - the deaths of 100,000 to as many as 250,000 seabirds, at least 2,800 sea otters, and many other animals • Long-term effects: more losses of species than expected • Bioremediation was somewhat effective in accelerating biodegradation as a cleanup option	Morris and Loughlin (1994); Boehm et al. (2014)
Pine River Oil Spill, British Columbia, Canada	Light crude oil	August 1, 2000	260,400	• 118,800 gallons removed from the river • Vacuum oil and set up containment booms of 12, 24 and 28 miles from the rupture • 119,560 gallons removed from soil • Removal of large woody debris from river channel due to it being coated with oil	• Extensive mortality of fish and some wildlife • Water supply to the District of Chetwynd was ceased and the use of many groundwater wells nearby the river was discontinued • Disruption of the river channel and destruction of fish habitat by seasonal flooding following the removal of woody debris that stabilized the river morphometry	Anderson and Schwab (2012); Chung et al. (2012)
Terra Nova Oil Spill, Newfoundland, Canada	Waxy crude	Nov. 2004	43,650	• 5% oil recovered	• An estimated 10,000-16,000 direct kills of birds	McGrath (2014)

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Wabamun Lake Oil Spill, Alberta, Canada	Heavy Bunker C fuel oil	August 3, 2005	343,000	• 185,000 gallons of Bunker C and Imperial Pole Treating Oil removed • Early response by sorbent booms (ineffective) • containment boom (inadequate quantity) • Later response by SCAT process and cleanup of individual shore segments based on geomorphology and degree of oiling • Extensive removal of shoreline oiled reed bed vegetation and mechanical recovery of oil from reed bed water column and sediments	• Residents were told to stop cleaning wildlife and to completely avoid use of all lake water and well water, even for watering gardens and lawns • Access to the lake for recreation and fishing was limited for months following the spill • Extensive damage to reed beds caused by oiled vegetation removal and recovery of oil and tar balls • Mortality of reed bed fish and wildlife, and reduction of nesting habitat for Western Grebe • Residual oil and tar balls in shoreline and marsh areas for several years • Evidence of oil contamination of whitefish spawning shoals at concentrations sufficient to impair embryo survival	Newbrey et al. (2012); Martin et al. (2014)
Deepwater Horizon Oil Spill, Gulf of Mexico, USA	Crude oil	20 April, 2010	210 million	• 33 million gallons recovered • Initially: ROVs, 700 workers, 4 airplanes and 32 vessels • Involved 47,000 people and 7,000 vessels in total • Chemical dispersants - Corexit (1.07 million gallons on water surface & 721,000 US gallons underwater) • *In situ* controlled burning (13 million US gallons) • Over 60 open water skimmers (33,000,000 US gallons or 120,000 m^3 of tainted water removed) • Extensive shoreline remediation, including shaping shorelines to direct oil away from sensitive areas and oil recovery from beaches and coastal wetlands	• Most impacts were on marine species – by November 2, 2010, 6,814 dead animals, including 6,104 birds, 609 sea turtles, 100 dolphins and other mammals • Actual number of mammal deaths due to the spill may be as much as 50 times higher than the number of recovered carcasses • Use of dispersant made oil sink faster and more deeply into the beaches, and possibly groundwater supplies • Beach erosion and disruption of plant and animal life-cycles were observed due to oil toxicity and cleanup actions	Allan et al. (2012); White et al. (2012); Barron (2012); NRC (2013)

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Kalamazoo River/ Enbridge Pipeline Spill, Michigan, USA	Cold Lake Blend (with benzene diluent)	July 26, 2010	819,000	• Over 1.1 million gallons of Line 6B oil recovered • Excavation and dredging	• Eighty miles of shoreline and adjacent lands were contaminated • Submerged oil was assessed and recovered at over 25 locations • Over 100 residents were relocated due to air quality (benzene) concerns • Residual oil in sediments producing sheen when disturbed • Extensive severe habitat damage and destruction due to removal of riparian vegetation, construction of roads on riparian lands up to the river banks, removal of islands that had been over-washed by oily water, and continual disturbance and removal of contaminated sediments	Dollhopf and Durno (2011); McGowan et al. (2012)
Lac-Mégantic Rail Disaster, Quebec, Canada	Bakken oil	July 6, 2013	26,455	• Removed oil and damaged rail cars from downtown Lac-Mégantic • Excavation of 57,250 m^3 contaminated soils • 49,494,000 L of oily water and oil were recovered and treated till December 19, 2013. • Run-off water management - oil and water recovery trenches and pumping stations • About 100,000 L spilled into the Chaudière River – shoreline cleanup	• Heavily contaminated with benzene causing heat and toxic conditions • Most-contaminated areas might never be habitable • 30 km stretch of the Chaudière River has residual oil in sediments and along the banks	Ecosocialist Network (2013); Lacoursière et al. (2015)
Gogama Oil Spill, Ontario, Canada	Synthetic crude oil	March 7, 2015	Over 264,000	• Containment booms installed • Skimmers and vacuum trucks deployed	• The spill contaminated a nearby lake, river and soil • The municipal water system was impacted by oil	Canadian National Railway Company (2015)

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7.2.6 Risk assessment

7.2.6.1 Spill risk assessment

Risk assessment underpins all preparation and planning for oil spill response and includes the assessment of both the likelihood of a spill occurring and the consequences or effects caused by a spill (Maritime New Zealand 2006). Substantial research has been done and progress made to develop analysis tools and evaluate risks of oil spills across the world and especially in many maritime countries, such as Canada, US, Norway, United Kingdom and Australia (Turner 2010). Refer to the discussion about oil spill risks and case studies in Chapter 8.

> The main components of a relative risk assessment of oil spills are:
> - Estimation of the probability of various sizes of spills based on past and projected future incident rates;
> - Vulnerability of the environment to oil spill impacts; and
> - Selection of the oil spill scenario(s) to be assessed; e.g., maximum most probable discharge and/or worst-case discharge.
>
> The assessments are conducted on a regional and seasonal basis by oil type. The level of detail used depends upon the spatial and temporal scope of the assessment and the availability of data.

WSP (2014a,b) have examined the potential frequency of spills in Canadian waters and the potential consequences associated with these spills. Potential spill frequencies were estimated using a combination of Canadian and worldwide spill statistics. Risk values generally increased in coastal areas and the overall highest risk was observed for small size spills due to their high frequency. Much of the conceptual foundation for these methods is provided by the Washington Compensation Schedule (Washington Administrative Code 173-183), which provides a relative impact score for oil spills, considering sensitivity of the locations oiled, relative density and seasonal distributions of sensitive biota, and factors related to oil type. The methods used in these reports are similar to those developed and demonstrated by Etkin (2012a), Reich et al. (2014) and Li et al. (2014a).

The general approach used by WSP (2014a,b) in their assessment of spills in Canadian coastal waters was as follows:

- Coastal waters were divided into four main sectors which were in turn divided into smaller subsectors for a maximum number of 77 zones;
- Shipping densities, as well as vessel types and size distribution, in each zone were estimated;
- Oil spill frequencies for ships were obtained from the most recent 10 years of worldwide data;
- The behaviour of oil spills (surface area over time) was estimated from simple transport and fate models which depend on the oil type, the spill size and location characteristics;
- The environmental sensitivity index (ESI) was calculated based on physical, biological and human metrics, which were further mapped to illustrate their spatial distribution in each zone; and
- The overall ESI was determined using a spreadsheet calculation and mapped to present its spatial distribution using GIS tools (**Figure 7.6**).

The key findings of the WSP (2014a) report on spills in southern Canada were as follows:

- The largest marine traffic volumes are in the Pacific sector where the probability of small size fuel spills is the highest. The zones with the highest probability of a large spill occurring were the waters around the southern tip of Vancouver Island, the Cabot Strait including southern Newfoundland, the eastern coast of Cape Breton Island and the Gulf of St. Lawrence and the St. Lawrence River;
- ESI results showed that the zones of highest potential impact were in the Estuary and Gulf of St. Lawrence, as well as in the southern coast of British Columbia including Vancouver Island. Higher ESI scores were observed in near-shore zones compared with intermediate and deep sea zones;

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- The highest relative risk values were for small spills due to their higher frequency of occurrence. The risk of large spills is generally low in Canada. The risk generally increases in near -shore zones compared with deep sea zones with the exception of the Pacific sector where US traffic may increase deep sea probabilities. The increase in risk in near -shore zo nes is related to an increase in environmental sensitivity; and
- The Estuary and Gulf of St. Lawrence, the St. Lawrence River, the southern coast of British Columbia, as well as three subsectors in the Atlantic sector , are at the greate st risk from large oi l spills. For the rest of the study area, the risk posed by spills over 10,000 m^3 were much lower. Risk is higher from small and medium spills in every sector, especially spills in the 100 to 999 m^3 range. These smaller spills can also cause significant damage and are likely to happen much more frequently than the larger spills.

These results can be used to tailor spill preparedness for each sector and subsector . *It would be useful, as a first step, to evaluate the current level of preparedness under the Ma rine Oil Spill Preparedness and Response Regime in light of relative risks in order to identify deficiencies.*

WSP used a similar methodology to assess the relative risk of oil spills in Canadian Arctic coastal waters (WSP 2014b). Physical Sensitivity Indic ators (shoreline type, wetlands, ice coverage), Biological Resource Indicators (ecologically and biologically significant areas and bird distribution and Human -Use Resource Indices (coastal population, tourism, national/international freight tonnage) were combined to produce an overall ESI for each of 18 subsectors. Spill frequencies were very low compared to southern Canada because of the current low amount of marine tanker traffic. The higher ESI scores occurred in the southern Arctic subsectors, including in the south of James Bay, the south of the Beaufort Sea and at the Mackenzie River Delta. Because the current probability of oil spills is low, relative risks were also low throughout all subsectors.

The WSP assessment of oil spills in Arctic waters did not include prediction of future risks due to increased traffic associated with the opening of routes due to climate change. The assessment also did not appear to account explicitly for the difficulty inherent in cleanup of Arctic spills, which would pote ntially produce more long-term risk.

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Figure 7.6 *Overall ERI for crude oil spills South of the 60th Parallel. Image from WSP (2014a)*

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Etkin (2012a) and Li et al. (2014a) proposed new modeling approaches to calculate the probability and/or vulnerability of oil spills by considering environmental conditions, shipping routes, incident cause, vessel type and/or nearby ecological reserves with uncertainties reflected in the system. The approaches were applied to the Cook Inlet vessel traffic and the southern Newfoundland offshore areas, respectively, where concerns of oil spill age exist due to shipping operations and potential impacts on the sen sitive environment of the area.

Recently, Reich et al. (2014) assessed the spill risk in Alaska/Arctic and identified the three highest relative risk regions within the scope of their stu dy for maximum probable spills and worst case spills as shown in **Table 7.6**. They pointed out that the relative risk model was highly data -intensive. Therefore, the quality of the results was dependent on the quality of input data. They cautioned that some environmental vulnerability data inputs were known to be of poor quality and should be updated. In particular, bottom habitat and submerged aquatic vegetation data coverage were lacking for much of the area assessed. Sensitivity testing of the model suggested that the number of species used for the study was sufficiently robust, but the addition of more species could refine the risk scoring. This would be important, for example, if species of high traditional use value to aboriginal people were a priority f or assessment. Reich et al. (2014) also pointed out that incident rates and potential spillage volumes forecasted for 2025 were subject to considerable uncertainty. Notwithstanding the limitations of the assessment, the results identified broad regions of Alaska with high relative risk, with implications for spill prevention and preparedness.

Table 7.6 *Summary of the relative risk rank and regions. Data from Reich et al. (2014)*

Relative Risk Rank	Max Probable Current Risk	Worst Case Current Risk	Max Probable 2025 Risk	Worst Case 2025 Risk
1	Southeast Alaska	Southeast Alaska	Beaufort Sea	Beaufort Sea
2	Aleutians	Kodiak/Shelikof Strait	Aleutians	Aleutians
3	Kodiak/Shelikof Strait	Cook Inlet	Southeast Alaska	Southeast Alaska

7.2.6.2 Environmental/ecological risk assessment

The general environmental/ecological risk calculation is expressed as the probability of spills multiplied with the potential impacts on environmental or ecological systems (US EPA 1999, 2004; NRC 2013). In general, the first step of risk assessment is hazard identification, which determines the qualitative nature of the potential adverse consequences caused by the action or contaminant. The second step is dose - response analysis, which determines the relationship between dose/frequency of action and the probability or the incidence of effect (dose -response assessment). The third step is exposure quantification, which measures the amount of a contaminant (dose) that individuals and populations will receive. And the la st step is risk management, including coordinated and economical application of resources to minimize, monitor and control the probability and/or impact of risks (Board ofOcean Studies 2014; Li 2014).

Many assessment methods and practices for oil spill ev ents have been reported. For example, French - McCay (2002) developed and validated an oil toxicity and exposure model (OilToxEx) for estimation of impacts to aquatic organisms resulting from acute exposure to spilled oil. The author further proposed a coupled oil fate and effects model for the estimation of impacts to habitats, wildlife and aquatic organisms resulting from acute exposure to spilled oil. The physical fate model estimated the distribution of oil (as mass and concentrations) on the water surfac e, on shorelines, in the water column and in the sediments. The biological effects model estimated exposure of biota of various behaviour types to floating oil and subsurface contamination, resulting percent mortality and sublethal effects on production (s omatic growth). Impacts were summarized as areas or volumes affected, percent of populations lost and

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production foregone because of a spill's effects (Liu et al. 2009). Recently, Etkin (2012b) developed a simple and defensible model to provide a relative quantitative measure of the environmental and socioeconomic impacts of a pipeline or facility spill as a function of key location-specific attributes. The model was intended to reflect significant differences in the sensitivity of different types of environments to spill-related damage without requiring a complex site-specific environmental impact assessment. Stephenson et al. (2015) studied hypothetical spills of dilbit in the Kitimat River and concluded that the potential for chronic ecological risk was the greatest for wildlife species occupying the floodplain or riparian habitat and having relatively small home-range sizes.

7.2.6.3 Health risk assessment

Health risk assessment is used to provide individuals with an evaluation of their health risks and quality of life. The main objectives include assessing human health, estimating the level of health risk and providing feedback to participants to motivate behavioural change to reduce risk. The results can reflect the impacts of oil spills and response activities on human health and therefore aid in decision-making as to whether or not and to what degree responses should be conducted (Crosby et al. 2013; Nicholls, et al. 2014; Bostrom et al. 2015).

Baars (2002) estimated the health risk for people involved in beach cleaning, sunbathing and swimming on the coast of Brittany, France, due to the wreckage of the oil tanker *Erika* in 1999. The results suggested that the risks were limited only to people who had bare-hand contact with the oil. Aguilera et al. (2010) reviewed the possible consequences of exposure to spilled oil on human health. Barenboim and Saveka (2012) introduced the biological impacts of individual petroleum hydrocarbons into the determination of the risk of oil spills for biota and humans. Michigan Department of Community Health (2014) evaluated the health risk associated with an oil spill on the Kalamazoo River and found that chemical levels found in surface water were not expected to cause long-term harm to people's health. It was also deemed that the concentrations of hydrocarbons found in fish from the Kalamazoo River and Morrow Lake would not harm people's health. Yee et al. (2015) carried out a Human Health Risk Assessment (HHRA) to assess both potential acute and chronic risks to human receptors in the unlikely event of a full bore pipeline break. The model was tested by a hypothetical spill with the proposed pipelines used for transporting oil extract from the Alberta oil sands to refineries or to coastal terminals for international export. The results showed that risk management strategies targeting the protection of the exposure pathways with relatively high contribution to human health risk (e.g., consumption of below ground and root vegetables and fish ingestion) during the remediation would be important to reduce the risk.

An emerging concern has been raised on the psychosocial impacts of a major oil spill. Grattan et al. (2011) used a community-based participatory model to conduct the standardized assessments of psychological distress (mood, anxiety), coping, resilience, neurocognition and perceived risk on residents of fishing communities who were indirectly impacted (n = 71, Franklin County, FL) or directly exposed (n = 23, Baldwin County, AL) to coastal oil. They found the residents of both communities displayed clinically significant depression and anxiety but no significant differences between community groups. They further pointed out that current estimates of human health impacts associated with the oil spill might underestimate the psychological impact in Gulf Coast communities that did not experience direct exposure to oil. Income loss after the spill might have a greater psychological health impact than the presence of oil on the immediately adjacent shoreline. Gail et al. (2014) compared the psychosocial impacts caused by the 2010 DWH oil spill and the 1989 EVOS based on surveys of local residents collected a certain period of time (12 and 18 months, respectively) after each of the spills. The analysis revealed similarly high levels of psychological stress, tied to renewable resources, concerns about their economic future, worries about air quality and safety issues regarding seafood harvests in oiled areas.

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It is valuable to incorporate a comprehensive examination of ecological and human health risks of oil spills into contingency planning and response decision -making (Cao and Fan 2013; Crosby et al. 2013; Bostrom et al. 2015). It would allow spill contingency planners and decision-makers to determine optimal risk-based response strategies, calculate the costs and benefits under different scenarios and choose the best scenario (or make the optimal decisions) to guide response actions (Gala et al. 2009; Etkin et al. 2011; Etkin 2012a; Fernandes et al. 2013). Linkov et al. (2006) suggested that decision analytical methods could incorporate risk assessment into integrated frameworks. A key attribute of the methods is the ability to engage stakeholders, providing for an opportunity for a shared understanding of the issues, and the importance of particular assumptions and model or data results on the assessment of alter natives (von Stackelberg 2013).

7.2.7 Cleanup process simulation and control

Process control is defined as an engineering discipline that deals with mechanisms and algorithms for maintaining the output of a specific engineering process within a desired range (Jing et al. 2012, 2015). *For oil spills, the knowledge and prediction of dynamic response processes to the variations of environmental conditions and operational factors are critical to ensure an optimal operation of the response process. A clear understanding of the mechanism of a response process (e.g., booming, in situ burning, skimming, dispersion and bioremediation) will help to quantify the direct relationships among the inputs (e.g., number and types of skimmers) and outputs (e.g., recovery rate), a s well as the indirect relationships, such as the time-series correlation* (Jing et al. 2015). Modeling of oil behaviour, effects and fate, along with the influence of spill response measures (e.g., skimming or dispersion), has been recognized as an essential component and foundatio n of successful process control strategies (Li et al. 2014b). *A numerically simulated response process can provide the decision -makers with a dynamic means to optimize oil spill recovery and cleanup operations. Particularly with the real-time aid of process simulation and control tools during spill response actions , the response efficiency and effectiveness can be promoted and the overall time and cost of recovery can be minimized.*

7.2.7.1 Gaps in cleanup process modeling

In dealing with an oil spill event, respon ders always require fast and accurate estimates of the spill to make timely and effective decisions in deploying skimmers, applying dispersants or conducting other response activities before, during and after the spill event. However, *complexity and dynamics inherently exist during oil spill response processes emanating from the fate and transport of spilled oil, existing climatic and oceanic conditions, operation of cleanup equipment, as well as their interactions, all of which cause significant challenges in response decision -making. Traditional physics -based spill models are weak in providing a good solution.* A few studies have explored the possibility of simulating oil recovery processes based on empirical oil weathering models and artificial equipment p erformance settings (Buist et al. 2011; El -Zahaby et al. 2011; You and Leyffer 2011; Zhong and You 2011), but *technical and knowledge gaps persist in how to obtain accurate and timely forecasting results under varying environmental conditions. This is espe cially true in harsh environments where the window of opportunity for spill response is significantly shorter, and more efficient decisions and actions are highly recommended.* Chen and his group (Jing et al. 2012; Li et al. 2012a, 2014b,c, 2015) proposed a set of simulation-based dynamic nonlinear programming approaches to optimize the number and operation of oil skimmers during a spill in a fast, real-time and cost-efficient manner based on the consideration and modeling of oil weathering processes. *Nonetheless, the modeling of response processes when using different cleanup techniques, such as booming, skimming, in situ burning, dispersant application and bioremediation, still face many challenges due to the lack of background data, high nonlinearity and various uncertainties from oil properties and weather conditions.*

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7.2.7.2 *Special considerations for harsh environments*

Cold and harsh environments in northern Canada and the Arctic are usually characterized by a wide range of wind speed and direction, limited visibility, low temperature, rough water surface, ice coverage, etc., posing substantial challenges for oil spill cleanup process simulation and control (Oskins and Bradley 2005; Keller and Clark 2008). For example, considerable amounts of fixed or floating ice may occur in the winter in both inland waters and offshore areas (Cleveland 2010). The presence of ice is a key factor affecting the ability to respond to a spill, as well as the modeling efforts (DeCola et al. 2006). The physical distribution and condition of spilled oil under, within or on top of the ice plays a critical role in determining the most effective response strategies at different stages in the ice growth and decay cycle. As pointed out by the National Energy Board of Canada's "State of Knowledge Review of Fate and Effect of Oil in the Arctic Marine Environment" (Lee et al., 2011), it is needed to understand the influence of these environmental variables/factors and the influence of various levels of ice coverage on oil fate and behaviours and the associated environmental impacts. A vast amount of basic information is available from the previous laboratory, mesocosm and field studies, as well as lessons learned from the past spills. However, *knowledge gaps still exit on the fate and behaviour of oil in water in ice conditions (solid, slush and frazil ice) and during active periods of formation and breakup of annual and multi-year ice, as well as the impacts on ecosystems. Consequently, the effects of applications of the different response methods under Arctic conditions are hard to forecast and manage in a more timely, eco-friendly and cost-effective manner.* The recent report by Arctic Oil Spill Response Technology Joint Industry Programme (AOSRT JIP 2014) provided a comprehensive review in this regard.

In-depth knowledge and modeling tools for oil fate and transport and effective response technologies are lacking for oil spills in ice and harsh conditions. Existing models have insufficient ability to accurately predict how oil and ice interact and then how oil transforms and moves where ice is present. Most mathematical equations used to simulate oil cleanup/recovery are likely based on empirical approximations and assumptions and are subject to time step and grid limitations. Furthermore, current response technologies are either ineffective or require more scientific and field validation in dealing with oil in ice conditions. These challenges should be addressed to support more effective response decision-making and operations.

<u>Recommendation:</u> Research is needed to develop physical and numerical models for simulating, predicting and optimizing response processes and evaluating their individual and collective effects on response decision and effectiveness, particularly in terms of accuracy and adaptability to different environmental conditions. The dynamic links with spill modeling and decision-making should also be realized.

7.2.8 *Response operation optimization*

The response to an oil spill is a dynamic, time-sensitive, multifaceted and complex process subject to various constraints and challenges. It is dependent on a variety of factors, including quantity and properties of the spilled oil, location, environmental conditions and availability and utilization of response resources at various degrees of oil weathering (Ornitz and Champ 2003). *Response operations usually need to be undertaken within a limited time window and improper decisions may compromise the efficiency of oil recovery and waste resources. It is highly desirable to develop and implement an optimized strategy to better coordinate different types of operations.* You and Leyffer (2011) developed an approach for oil spill response planning with integration of the physiochemical evolution of oil slicks. The approach includes a dynamic oil weathering model and can simultaneously predict the optimal coastal protection plans and oil spill cleanup schedules with different types of mechanical, burning and dispersant application equipment. The results demonstrated the importance of integrating an oil transport and weathering model in response planning. Zhong and You (2011) addressed the optimal planning of oil spill response operations under the constraints of economic and responsive criteria, with consideration of

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oil transport and weathering processes. A multi-criteria, multi-period linear programming model was developed that minimizes the total cost and response time-span and simultaneously predicts the optimal time trajectories of the oil slick's volume and area, transportation profile, usage levels of response resources, oil spill cleanup schedule, and coastal protection plan. Li et al. (2012a, 2014c) proposed a simulation-based nonlinear optimization approach to provide sound decisions for skimming spilled oil in a fast, dynamic and cost-efficient manner, which especially targeted harsh marine environments. The model was further integrated with the optimization method to determine the optimal strategy to achieve the maximum oil recovery within constraints of time and resources. The approach was tested using hypothetical case studies showing its capability of efficiently incorporating the process simulation and optimization into the same modeling framework to improve skimming effectiveness. Efforts are being made to incorporate these approaches into existing modeling software or new decision systems with user-friendly interfaces to become effective tools for onsite use and training by responders.

Recommendation: In general, response decision-support systems are rare and only present limited degrees of integration between oil spill modeling, cleanup process simulation and/or risk assessment, but their dynamic and interactive features need to be significantly enhanced, and more field validation efforts are urgently required to increase the confidence for implementation.

Uncertainty, which is one of the major hindrances to improving the efficiency of cleanup process simulation and control, may arise from a variety of sources. Such sources include, but are not limited to, incomplete information, errors in sampling, subjective judgment, random variations of and dynamic interactions among operating factors, approximations and assumptions in measurement, and changes of environmental conditions (Huang et al. 2001; Lin et al. 2009; Liu et al. 2009). *Uncertainties lead to difficulties in developing optimization models for supporting decision-making in oil spill response and impair confidence of decisions.*

7.2.9 Information technologies for response decision making

7.2.9.1 Geomatics, remote sensing, and geographical information systems

Geomatics (or geomatic technology or geomatic engineering) is a hybrid field of gathering, storing, processing and presenting geographic information or spatially referenced information (Zhu et al. 2013; Song et al. 2014). The techniques of geomatic analysis, remote sensing and GIS have been widely used in oil spill response. The Central Gulf of Mexico Ocean Observing System (CenGOOS), along with remote sensing and GIS, were adopted in monitoring and forecasting the DWH spill (Howden et al. 2011; Li et al. 2012b). Morović and Ivanov (2011) discussed the properties of synthetic aperture radar (SAR) imagery as the most reliable source of oil spill information and the possibilities of a combined SAR-GIS approach for oil spill monitoring and management for the protection of the Adriatic Sea. Pan et al. (2012) used multi-source remote sensing data, including MODIS data, and advanced SAR images to study the aftermath and delineate the extent of the 2006 Lebanon oil spill. Dollhopf et al. (2014) introduced a multiple-lines-of-evidence approach for assessment and recovery of submerged oil by integrating with a robust GIS tool. The approach helped optimize the assessment and recovery, inform the decision-making related to transitions and endpoints, and improve the cleanup efficiency while limiting the long-term ecological damages from the recovery activities.

7.2.9.2 Data mining and artificial intelligence

Knowledge Discovery (KD) effectively uncovers hidden but subtle patterns from large and diverse datasets and outperforms traditional statistical techniques. Data mining, a major stage in the KD process, is the analysis of datasets that are observational, aiming at finding out hidden relationships among datasets and summarizing the data in a manner that is both understandable and useful to the users. Many artificial intelligence (AI) tools have been used for data mining, such as neural networks, fuzzy logic, ant

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colony algorithms and genetic algorithms. *Given these features, data mining and AI tools have received increased interest and efforts in addressing the complexity and dynamics of oil spill and response operations and seeking optimal solutions for risk assessment and decision -making* (Cai et al. 2009; Akinyokun and Inyang 2013). Some new methods, such as fuzzy neural network (Cao and Fan 2013; Inyang and Akinyokun 2014) and predictive Bayesian model (Echavarria -Gregory and Englehardt 2015), have been developed and applied to assess oil spill risks and characterize their patterns.

7.2.9.3 Results visualization

Preferably, an integrated oil spill response decision support system would include visualization of not only past and current situations, but a dynamic probability envelope for the future spill positions and the associated risks, time and costs under different response scenarios, much as is done in the hurricane/typhoon and flooding forecasting community (Sanyal et al. 2010). However, *present oil spill visualization systems are generally based on producing either a snapshot map of representative oil spill particle trajectories or a movie of an evolving point cloud of particles* (Kang et al. 2013; Hou and Hodges 2014). Furthermore, as geomatic technologies become more powerful and usable over desktops and even mobile platforms (e.g., smart phones, tablets), oil spill visualization systems should use GIS standard formats for output data to allow web -based access for emergency response personnel (Yu and Yin 201 1). *Standardization within a GIS can allow spill trajectories to be linked to existing index systems (e.g., environmental or ecological sensitivity index) that classify sensitive coastal areas by their degree of exposure and vulnerability.*

Fernandes et al. (2013) described recent updates of the oil and inert spill modeling component of MOHID model, including interfacing with meteorological and oceanographic data and the integration with decision support systems. This system revealed a good stability and str ong performance in visualization of weathering processes and properties. The rich image resources of satellite remote sensing Google Earth was selected as the client during constructing the system. The system solved data exchange and data storage of KML be tween Google Earth and Oracle by connecting them, which realized the visualization of the oil spill system. Hodges et al. (2015) introduced methods to integrate existing oil spill models, servers, connections to online data services and visualization tools to support response decision -making. To improve deepwater oil spill emergency response, Wei et al. (2015) built a high -resolution hydrodynamic data field and a 3D visualization system based on oil behaviour and fate.

The oil spill response community will greatly benefit when current technical challenges and knowledge gaps are overcome to enable the coupling of advanced modeling and visualization techniques to address questions such as:

- How to integrate a dynamic spill modeling system with remote sensing, GPS and GIS to turn real-time information into real-time decisions and then visualize them;
- How to combine oil fate and transport modeling, environmental risk/impact assessment and cleanup process simulation and optimization with information technology to achieve the best response efficiency while minimizing time and costs; and
- How to visualize the levels of uncertainties within models to evaluate their risk.

7.3 Summary

This chapter reviewed the current development and practice of oil spill prevention and re sponse decision-making in Canada and beyond. Prevention and preparedness have been emphasized as the key factors to proactively reduce the potential damage of oil spills. With a high level of knowledge, many potentially high-risk activities can be anticipated and may be mitigated or prevented. Cutting-edge monitoring and oil analysis technologies, as well as advanced vulnerability assessment and contingency planning methods, can help identify and even prevent unexpected oil spills in an efficient, timely ma nner. Baseline data of

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environmental and ecological conditions before spills take place in areas of high spill probability are critical in support of spill preparedness and response decision-making. A well-structured response regime, an established spill c ontingency plan and existence of active programs for training, drills and education are critical for a country or region to effectively combat oil spills and minimize the damage to the environment and danger to human health.

Responses to an oil spill can b e highly dynamic, time -sensitive, multifaceted and complex due to uncertain or changing information (location, oil properties, weather, currents, etc.) and response resources (devices, manpower, money, etc.). A successful response operation usually starts with an accurate detection of an oil spill through imaging analysis and early warning. The selection of BATs and the precise prediction of oil trajectory to combat the oil spill also play important roles in terms of saving cost and time. NEBA and risk asse ssment tools usually offer the decision -makers quantitative information to evaluate the damage and to prioritize the response technologies. Cleanup processes should be simulated and optimized simultaneously in order to improve the response effectiveness wi thin a limited time window. With the aid of advanced techniques, such as geomatic analysis, AI and computer visualization, oil spill response decision support systems should be capable of not only integrating spill and cleanup modeling and risk assessment but recommending, and graphically presenting in the most simplistic format, optimal decisions and corresponding impacts under different scenarios to responders and stakeholder before or during a spill. In addition, post -event assessment is recommended to d eal with changing conditions, such as weather and subsequent work activities. It should be noted that the efforts of integrated oil spill response decision -making have been limited due to the complex, dynamic, and uncertain nature of oil spills. More efforts toward the improvement of oil spill preparedness and decision-making are highly desirable, particularly for some emerging challenges such as dilbit spills, derai lments and harsh environments.

7.4 Research Needs and Recommendations

7.4.1 High Priority Research Needs

- Research is needed to develop modeling methods to simulate, predict and optimize individual and collective cleanup processes (e.g., booming, in situ burning, skimming, dispersion and bioremediation) and their effects individually and collec tively for supporting response decision making.
- Research is urgently required on development and demonstration of integrated oil spill response decision support systems (DSS). Such systems should be capable to integrate monitoring and early warning, spill modeling, vulnerability/risk analysis, response process simulation/control, system optimization, and visualization in a dynamic and interactive manner. Response decision making and implementation are challenged by harsh environments in northern regions an d Arctic waters which has a short window of opportunity and can pose large amounts of uncertainty and difficulties in using traditional modeling and response methods.
- Research investment is needed for trial tests and field validation of new prevention and decision-making methods to demonstrate feasibility, increase confidence for implementation and improve response capabilities.
- Research is needed to better quantify uncertainties, evaluate their propagation and mitigate their impacts throughout spill resp onse systems and decision -making processes. Uncertainty is one of the major hindrances to improving the efficiency and integrity of spill modeling and decision making tools and then the confidence and effectiveness of response decisions.
- Further research is desired on environmental forensics (e.g., new biomarkers and fingerprinting methods), remote sensing and in situ measurement, and biological monitoring to improve spill prevention and decision-making.

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- Special research efforts should be given to the harsh and Arctic environments and some other emerging concerns (e.g., diluted bitumen, aging/subsea pipelines, and railcars) for improving effectiveness and confidence of prevention and response strategies and decisions.

7.4.2 Medium Priority Research Needs

- Concerns over acquisition, integrity, sharing and uncertainties of baseline data make oil spill contingency planning and response decision-making less able to meet response needs. (refer to Chapter 4)
- Innovative research should be strongly encouraged to develop more cost-efficient and eco-friendly response technologies (refer to Chapter 6). Meanwhile, there is a need for updated regulations to govern the use of response technologies (e.g., dispersants) especially for their Arctic and subsea applications, which should be based on improved knowledge and better understanding of their long-term impacts.
- Oil leakage from routine operations and discharges of hydrocarbons with wastewater (e.g., produced water, ballast water, bilge water and quench water) in the oil and gas industry is not given sufficient attention in comparison with accidental spills in terms of research and policy development. Quantitative evaluation of cumulative effects of long-term leakage and/or discharge of persistent and toxic oil components (e.g., PAHs) and development of in situ advanced treatment and recovery methods are recommended.
- Most response systems lack effective early warning functions/ability to identify, diagnose and react promptly to minimize oil discharge into the environment at the early and critical stage of the emergency. Incorporation of early warning indicators for potential events into real-time monitoring, quantitative risk analysis and response preparedness is recommended.
- Oil leak detection systems need to be further developed to detect spills rapidly after an incident. More comprehensive and reliable in situ and remote detection methods are needed. Research and implementation of autonomous, remotely operated detection devices (e.g., AUVs and ROVs) technologies should be encouraged.

7.4.3 Low Priority Research Needs

- Analysis of cumulative impacts remain a challenge for NEBA of oil spill response options. Influence of human judgment and socioeconomic factors should be more effectively reflected in the NEBA process. There is also a need to assess the NEBA against emerging issues of concerns (e.g., dilbit), countermeasures (e.g., bio-dispersants) and special environmental conditions (e.g., ice coverage, permafrost and peat lands) to identify knowledge gaps for decision making.
- It is critical to properly assess the real significance of spills and subsequent response activities and to develop lessons learned in preparation for future events. The requirements for improved onsite and post-incident monitoring and assessment should be better defined. Adaptive management, as a systematic process, is encouraged to continually improve management policies and practices by adjusting subsequent actions based on learnings from past events and new knowledge development.
- There is no comprehensive national framework in place for training and drilling exercises for ship-source oil spill preparedness and response that involves all key stakeholders. Most current exercises are artificial as the equipment and vessels are pre-assigned and the personnel are pre-notified or even pre-positioned. Unannounced spill response drills are sometimes necessary to test the operator's response capacity to contact, move and deploy the whole response team.
- Education through social media is being used and should be further encouraged in order to improve the way the community is informed and engaged during oil spills about the situation and how decisions are made. Educational outreach to enhance literacy and promote preparedness and responses should be a key part of the training..

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CHAPTER 8: RISKS FROM OIL SPILLS

Abstract

This chapter provides a review of the important factors affecting the consequences of selected oil spill case studies and presents lessons learned from each spill. Uncertainties and research needs revealed by each spill case are presented. The strengths and weaknesses of relative risk assessments are discussed and recommendations are made to improve the current practice of risk assessment in Canada.

A review of selected oil spill cases illustrated that each case had a unique combination of site-specific factors. There were different combinations of factors that either increased or decreased the overall impacts of each spill. The conclusions arising out of examination of the case studies are listed below.

- Delayed response was a critical factor affecting the consequences of all the oil spill case studies. Notwithstanding the importance of weather, remote locations and technological challenges, human error (at an individual and organizational level) was a dominant factor across all case studies. Absent or inadequate planning, insufficient integration, inadequate training, poor communication, insufficient capacity (personnel and equipment), poor or no information sharing, and lapses in regulatory oversight were noted for most, if not all, spill case studies.
- The consequences of oil spills cannot be predicted simply on the basis of the type of crude oil, although in general, spills of light crude had greater acute effects. Long-term effects were dependent upon a combination of the type of crude oil, characteristics of the receiving environment (including season) and effectiveness of cleanup.
- The lack of pre-spill baseline data seriously limits the ability to predict or monitor long-term effects of crude oil spills on populations and communities of aquatic organisms.
- Major advances in the development and validation of spill response technologies are seriously limited by the inability to conduct controlled field experiments with oil.
- Standard accepted methods have not been established for post-spill monitoring and validating the efficacy of oil spill response protocols and operational endpoints (How clean is clean?).
- Effects of oil spill cleanup on aquatic ecosystems can be significant.
- Aquatic ecosystem resilience to shocks from oil spills is the key to the maintenance of ecosystem services. The presence of long-term stresses may tax the capacity of ecosystems to be resilient to shorter-term shocks.

A review of relative risk assessments for coastal areas of Canada and Alaska and risk assessments done in support of Environmental Impact Assessments (EIAs) generated the following conclusions:

- The data needed for input to risk assessments in Canada are often either absent or widely scattered among government agency, industry and academic sources;
- Information needed for reliable environmental sensitivity indices is very limited for large portions of Canada;
- Input from Indigenous peoples and interested parties is required to produce reliable and credible information to be used for environmental sensitivity indices;
- Assessments of the risks of hypothetical pipeline spills are likely underestimated given the assumption that it would take only a few minutes to recognize a pipeline breach and deploy shut-off valves; and
- The level of sophistication of risk assessments conducted in support of EIAs varied substantially between regions.

High-priority recommendations arising out of these conclusions are:

- Collect and evaluate baseline information from high-risk, poorly-characterized areas in Arctic coastal areas and in freshwater systems (e.g. inland rivers with multiple pipeline crossings, the Great Lakes, the Gulf of St. Lawrence);
- Conduct controlled field experiments on oil spills at a variety of sites for the development and verification of current and emerging spill response countermeasures;
- Conduct long-term research into effects of different crude oil types on populations of aquatic biota, including follow-up research at the site of old spills;
- Conduct a program of research on resilience of aquatic ecosystems;
- Develop national guidance for monitoring of oil spills to ensure that information gathered is reliable, adequate, credible and consistent. The guidance should include provisions for adjustment in response to specific characteristics of the receiving environment and should provide details on operational endpoints which identify when cleanup and remediation can cease;
- Include studies of the relative effectiveness of response measures in all investigations of significant oil spills in Canada;
- Pre-approved funding should be put aside for research and monitoring of 'spills of opportunity';
- To support the development of improved spill response guidelines, build a comprehensive national database for the fate, behaviour and effects of various types of oil spilled and the efficacy of current and emerging oil spill countermeasures over a range of environmental conditions;
- Investigate the socioeconomic impacts of oil spills in support of assessment of effects on ecosystem services;
- Conduct regional risk assessments in areas of concern to provide further guidance for policy and planning;
- Establish a 'standard of practice' regarding the assessment of risk of oil spills in support of EIA in Canada which includes updated and refined assessment methods and a re-examination of appropriate spill scenarios; and
- Integrate traditional knowledge and information about traditional uses of resources into development of sensitivity indices used for risk assessment.

Introduction

To cover the range of potential spill scenarios that Canada may encounter in the future, this chapter examines the risks arising from oil spills into aqueous environments in two main sub-sections: oil spill case studies and reviews of risk assessments of hypothetical spills.

The case studies clearly illustrate that each spill incident is a unique combination of site-specific factors. These factors include: the type of oil; the mechanism of release; the volume of the spill; the weather/hydrology conditions at the time of the spill; the vulnerability of the habitats; the sensitivity of the biota; and the effectiveness of applicable spill response options. The impact of a spill is often influenced by the degree to which the oil reaches shorelines and sediments, as well as the physical-chemical characteristics of the oil (e.g., tar balls, liquids trapped under cobbles, oil mixed with sand, 'pavement', etc.). The interactions of various physical-chemical forms of oil with shorelines and sediments influence environmental persistence, bioavailability, toxicity and net environmental benefit of countermeasure strategies applied. In each of the case studies and risk assessments reviewed in this chapter, different combinations of factors either increased or decreased the overall impacts.

The case studies were selected to present combinations of geography, oil type and environmental factors that influenced the response, outcome and knowledge gained:

- Marine environments:

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- The *Arrow* tanker spill of heavy fuel oil in Chedabucto Bay, NS.
- The Baffin Island Oil Spill (BIOS) Project involving deliberate release of crude oil into the Arctic environment.[1]
- The *Exxon Valdez* tanker oil spill (EVOS) of medium crude oil into cold, pristine conditions with intensive shoreline intervention, in Prince William Sound, AK.
- The Deepwater Horizon (DWH) spill of light crude into the deep subsurface and shoreline environments of the chronically oil-impacted Gulf of Mexico, with unprecedented subsurface application of dispersant.

- Freshwater environments:
  - A pipeline spill of sour light crude into the Pine River, BC.
  - A derailment spill of a type of Bunker C oil (HFO 7102) into Wabamun Lake, AB.
  - A pipeline spill of dilbit into Talmadge Creek and the Kalamazoo River, MI.
- Recent Canadian spills in 2015:
  - The *Marathassa* Intermediate Fuel Oil (IFO) spill in English Bay, Vancouver, BC.
  - Nexen Long Lake pipeline bitumen emulsion spill near Ft. McMurray, AB.

For each case study, the important factors affecting the consequences are identified. Lessons learned are then listed. Finally, a list of research needs illustrated by the case study are identified and presented in **bold**.

The risk subsection reviews: 1) predictive relative risk assessments conducted in support of policy and planning (Transport Canada's review of Canada's preparedness and response for ship-source spills); 2) risk assessment in support of EIAs prepared as part of the regulatory approval process for proposed projects (e.g. the Northern Gateway Pipeline); and 3) risks from crude-by-rail transport. Reviews included evaluation of the risk assessment methods used, the key factors contributing to risk, and the primary uncertainties associated with the prediction of risk from oil spills. Recommendations for the improvement of risk assessment based on the Panel's findings are presented in **bold**.

8.1 The Arrow Spill

On February 4, 1970, the aging tanker *Arrow* encountered severe weather as it entered Chedabucto Bay, NS. She ran aground on Cerberus Rock, a well-known navigational hazard. Severe weather hampered attempts to off-load her cargo of Bunker C crude oil (Table 2.2) and on February 8, the tanker split into two releasing about two-thirds of her cargo (10,000 tonnes of oil) into the icy waters of Chedabucto Bay, much of which took the form of an oil-in-water emulsion that affected 300 km of coastline. Retrieval of the remaining oil from the wreck was completed on April 11, 2015, using pioneered techniques subsequently used in other tanker accidents (Maritime Museum of the Atlantic 2015).

The *Arrow* spill provides a long-term case study of reliance on natural attenuation (defined in Chapter 3). Only 48 km of the approximately 305 km of oiled shoreline were cleaned (Owens 2010). Cleanup primarily involved manual and mechanical removal of oiled coarse sediments, typically contaminated to a depth of 25-100 cm and occasionally as much as 200 cm (**Figure 8.1**). The average volume of oiled sediment generated by manual cleanup was approximately an order of magnitude less than mechanical removal using bulldozers and front-end loaders; this had implications for waste management. In terms of long-term impacts, mechanical removal of oiled beach material resulted in retreat of the beach crest at some locations for periods up to 10 years (Owens 2010).

[1] *In addition to the information summarized in this report, the Panel notes a recent report by the Arctic Oil Spill Response Technology Joint Industry Programme (AOSRT 2014), which provides a comprehensive literature review and recommendations regarding the assessment of environmental impacts of Arctic oil spills.*

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Figure 8.1. Testing the use of lime during shoreline clean-up of the Arrow spill. *Image courtesy of Ed Owens.*

Time-series measurements of petroleum -derived hydrocarbons in the water column of Chedabucto Bay demonstrated that concentrations returned to pre -spill levels within a year (Levy 1972). Time-series measurements in 1970, 1973 and 1976 showed a dramat ic reduction in shoreline oiling, with 75% of the heavily oiled shoreline cleansed by 1973, even in sheltered areas with low wave energy (Owens 2010). By 1992, approximately 250 km of oiled shoreline were cleaned naturally, of which about 70 km were sheltered environments where physical processes such as wave action and abrasion would have played only a minor role in weathering and degradation (Ow ens et al. 1993; Owens 2010). The coastal waters of northern Chedabucto Bay are rich in clays and these clays ca n interact, albeit slowly, with shoreline oil suggesting that the formation of clay -oil emulsions (also referred to as clay-oil flocculation, oil-mineral aggregates or OMA (Chapters 2 and 6) played a significant role in the natural removal of shoreline oiland subsequent biodegradation (Owens 2010).

Twenty years after the spill, Vandermeulen and Singh (1994) studied two beach sites in Chedabucto Bay and found that both contained a spectrum of degraded petroleum residues, from seemingly unaltered to severely weathered residues. The authors noted that the relatively permeable nature of the Chedabucto Bay sediments permitted stranded oil to become sequestered and protected within shore sediments (**Figure 8. 2**). The authors concluded that persistence was a direct function of beach sediment permeability, sediment grain size and the depth to which entrapped tar residues penetrate. Persistence was an inverse function of the depth and frequency in which sediments were reworked during tidal incursions. The authors noted that the oil residues trapped within the spaces of protected sand -gravel lagoons and deep cobble/boulder beaches would be the primary long -term source of hydrocarbon movement to pore water and surface water. The chemistry of some of the sequestered residu es still closely resembled the original spilled oil (Vandermeulen and Singh 1994). In contrast, the surface -stranded Bunker C residues progressively weathered to tar via exposure to oxidation, photooxidation (Chapters 2 and 6) and physical weathering by wave action, sediment scouring and tidal washing.

Thirty-five years later, most of Chedabucto Bay was oil -free, but the areas with surface oil distribution had changed little since mapping in 1992, and some areas of Black Duck Cove (a low -wave energy site) still had residues on coarse -sediment beaches and within fine -grained sediments at an adjacent protected lagoon (**Figure 8.2**, Owens et al. 2008). Exposed highly biodegraded and photooxidized surface asphalt pavements observed in the upper intertidal and sup ratidal zones in 1992 were being eroded by wave action at slow but observable rates. Subsurface oil that filled the pores below the boulder surface layer was associated with the presence of finer sediments, limiting the downward migration of the oil. Analysis of samples from these environments showed that the oil close to the surface was extensively degraded relative to subsurface oil recovered at depths beneath the surface layer of boulders, cobble and fine sediment where the composition of residual oil wa s similar to the original spilled material. The authors

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suggested that the more deeply buried residues in the cobble - boulder areas would likely remain until the sediment was disturbed by major storms as a result of landward barrier migration. These observa tions on oil persistence supported those from a previous study in the 1990s (Vandermeulen and Singh 1994). Owens et al. (2008) noted that residual oil in the fine-grained sediment of the lagoon area of Black Duck Cove was extensively biodegraded. The authors suggested that this indicated that if unweathered oil is released from the cobble areas into the lagoon, it would be rapidly degraded.

High wave energy may strand oil in the supratidal zone but also removes the oil on shores by erosion. A hurricane removed a sandy beach in Nova Scotia being studied by researchers, and winter storms may have been the most effective mechanism for the removal of stranded oil on rocky shorelines following the EVOS (K.Lee, pers. comm.)

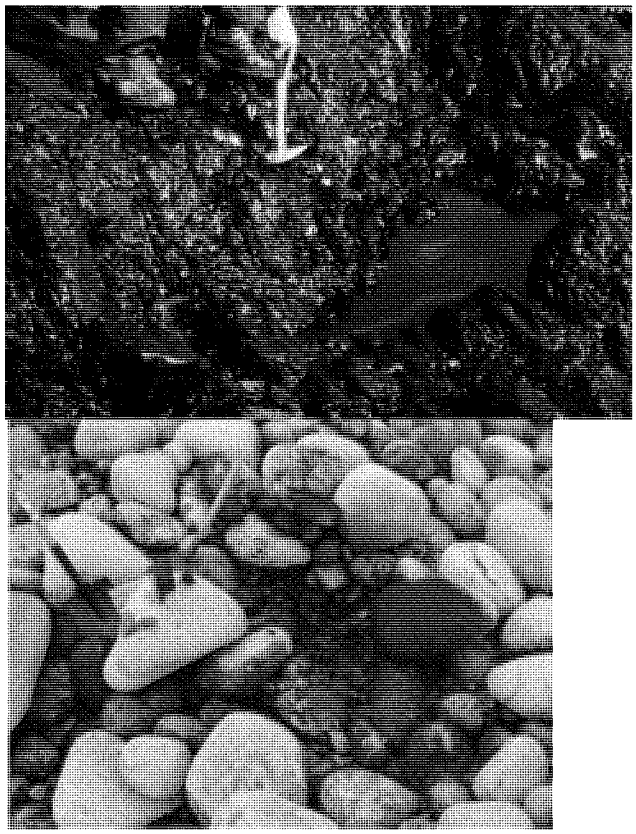

Figure 8.2 Comparison of oil on the shoreline immediately after the Arrow oil spill in Chedabucto Bay, N S and thirty-five years later *. Left: Cobble/boulder beach in 1970 immediately after the spill . Image courtesy of DFO. Right: Interstitial oil residue in Black Duck Cove in 2005. Image from Owens et al. (2008)*

The shorter-term effects of the *Arrow* spill were acute fauna mortality from exposure to the more toxic (and shorter-lived) components of the oil , as well as physical smothering of waterfowl. Certain deposit-feeding polychaetes had a high tolerance for living in sediments contaminated with the Bunker C oil , and their feeding activity accelerated the weathering rate of oil through various processes , including bioturbation and digestion (Gordon et al. 19 78a). Other intertidal organisms appeared to be more susceptible. Thomas (1978) observed that within the first six years after the spill, effects included extensive mortalities of the attached algae *Fucus* on rocky shores and mortality of clams and marsh grasses in lagoons. Species diversity was generally lower at oiled sites.

The longer-term ecological risks of the *Arrow* were largely a function of the weathering and release of hydrocarbons from the oil sequestered within beach materials. Longer -term effects were investigated by Lee et al. (1999) in 1993 and 1997, when oil was evident as asphalt pavements, stain on rocks, surface

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sheen and liquids trapped within sediments at some low -energy sites, such as the lagoon in Black Duck Cove.

Lee et al. (1999) conducted tests at a representative site in Black Duck Cove to determine the toxicity of residual oil to fish (winter flounder), invertebrates (amphipods, sea urchins and grass shrimp) and bacteria. They found that while much oil remained in the sediment, it was of low toxicity in most cases (except for significant mortality in an amphipod survival test) as it had undergone significant biodegradation (Lee et al.1999) and photooxidation processes (Prince et al. 2003). Habitat recovery was indicated by field observations of diverse benthic invertebrate communities within the sampling area (Lee et al. 2003), including green crabs (*Carinus maenus*), soft -shelled clams (*Mya arenaria*), common periwinkles (*Littorina littorea*), amphipods (e.g. *Gammarus oceanicus*) and worms (e.g. *Nereis* sp., *Arenicola* sp.). Exposure to elevated concentrations of bulk remaining oil within the sediment at the site to the flora and fauna has been reduced by burial at depth and dilution by storm events over the years. Tissue hydrocarbon concentrations in *Mya* had declined to levels where residents were harvesting clams from the study area, part of which had become a public park.

Wang et al. (1994) noted that after 22 years of weathering in the environment, most saturated hydrocarbons and polycyclic aromatic hydrocarbons (PAHs) in *Arrow* oil samples had been lost. The analysis of biomarker compounds by GC/MS were required to link residual oil samples to the *Arrow* cargo oil as traditional fingerprint methods could not provide detailed information of the source, characteristics and fate of the spilled oil.

In conclusion, the *Arrow* spill provides a valuable example of the fate and effects of spilled oil and the relative merits of cleanup versus natural attenuation for heavily oiled coarse-grained beaches in mid-latitude environments. Natural processes, including the interaction of oil and fine mineral particles that enhanced the physical dispersion processes, were effective in significantly reducing shoreline oiling such that residual risks appear to have been minimal.

8.1.1 Important factors affecting the consequences of the Arrow spill

8.1.1.1 Oil properties and behaviour

- Severe weather and turbulence resulted in both rapid dispersion of the oil and deposition onto shorelines and lagoon areas.
- Formation of OMAs contributed to relatively rapid natural attenuation of shoreline oiling

8.1.1.2 Effect of the environment on fate and behaviour

- The presence of cobble/boulder beaches and lagoons with fine sediments provided areas where oil was sequestered and persisted.
- Storms buried and then re-exposed some of the spilled oil.
- Oil under aerobic surface conditions biodegraded at a much faster rate than oil sequestered within sediments with limited oxygen availability and water/nutrient exchange.
- Bioturbation by benthic organisms enhanced oil biodegradation rates.

8.1.1.3 Oil toxicity

- Acute mortality was associated with exposure to the more toxic (and shorter -lived) components of the oil, as well as physical smothering.
- Reductions in chronic effects were correlated to the weathering of the residual oil (including biodegradation) and the episodic release of buried oil within beaches as the result of storm events.

8.1.1.4 Spill response

- The application of active spill response operations was limited by the lack of availability of spill response measures and logistic constraints linked to inclement weather conditions.
- Mechanical removal of oiled sediments generated up to 10 times more solid waste than manual removal.
- Removal of large volumes of coarse-grained sediment without replacement resulted in beach retreat.
- Oil was naturally removed in low-wave-energy environments where fine particles created OMAs that enhanced physical dispersion and biodegradation of the oil.
- The persistence of residual oil was generally inversely correlated to shoreline energy level (e.g. from waves, tides and ice) with oil remaining longer in low energy environments.

8.1.2 Lessons learned from the Arrow spill

- Delayed response due to adverse weather and oil spill response preparedness increased the volume (due to water-in-oil emulsification processes) and extent of spilled oil.
- Over time, natural attenuation can effectively mitigate the environmental impacts caused by oil spills in the marine environment.
- Socioeconomic effects can be both short- and long-term; short-term effects include loss of income among fishermen. Long-term effects are largely aesthetic, and these can cause loss of amenities (e.g., beach use) and tourist income.

8.1.3 Research needs related to the Arrow spill

Recommendation: Further assessment of cumulative effects of the residual oil in Chedabucto Bay should be conducted. The assessment should evaluate species and indicators relevant to all stressors of concern (from all sources) within a defined study area appropriate to the current and predicted future location of stressor sources. The assessment should establish clear thresholds of effect for each indicator such that the incremental contribution of the residual oil to overall cumulative risk can be better understood.

Recommendation: An assessment of the residual effects of the *Arrow* spill on ecosystem services would provide useful information for incorporation into net environmental benefit analysis (NEBA), as well as input to future risk assessments.

8.2 The Baffin Island Oil Spill Project (BIOS)

The BIOS project was conducted from May 1980 to August 1983 in the eastern Arctic at Cape Hatt, on the northern end of Baffin Island (Sergy and Blackall 1987). Information from the study and others, such as the Balaena Bay study [2] were intended for planning, approval and operational phases of hydrocarbon development in the Arctic (Lee et al. 2011).

The BIOS Project consisted of a shoreline study and a near-shore study that involved controlled releases of crude oil. Its primary objective was to determine if the use of dispersants in the Arctic near-shore would reduce or increase the environmental effects of spilled oil and to determine the relative effectiveness of other shoreline protection and cleanup techniques. The secondary objective was to determine the chemical and physical fate of oil in Arctic near-shore and shoreline areas (Sergy and Blackall 1987).

[2] *Conducted in 1974-1975 with the principal objectives of assessing the impact of an offshore oil well blowout on the thermal regime of the Beaufort Sea and developing potential countermeasure techniques (NORCOR Engineering Research Ltd. 1975).*

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The shoreline of the study area consisted of steep, rocky promontories separating coarse sand, gravel and cobble beaches. Open water occurred for about 65 days per year. Tides ranged from 1 to 2 m and the mid-summer mean maximum air temperature was about 7 °C (Sergy and Blackall 1987). Pre-spill analysis of water, sediment and benthic fauna confirmed that there were very low background petrogenic hydrocarbon concentrations (Cretney et al. 1987a, b, c).

Three bays of similar coastal morphology and sedimentology were selected as test areas. Oil fate was monitored in the intertidal and shallow subtidal zone in the water column, intertidal beach sediments, subtidal sediments and the tissue of selected benthic invertebrates (Boehm et al. 1987; Humphrey et al. 1987a, b; Owens et al. 1987). The measurement of biological effects focused on the small and less mobile subtidal benthic flora and fauna (Sergy and Blackall 1987). The experimental approach was not suitable for the direct measurements of effects on fish, sea birds and marine mammals.

8.2.1 The near-shore study

The near-shore study compared the consequences of dispersing an oil slick close to shore with the consequences of allowing the oil to beach and leaving it to natural attenuation. Two experimental discharges of 15 m^3 of weathered (8% by weight) sweet medium crude oil (Venezuela Lagomedio) were released at each site over a period of six hours or half a tide cycle. Corexit 9527 was the dispersant used in a volumetric dispersant/oil ratio (DOR) of 1:10 (Lee et al. 2011).

The surface slick release began at high tide and under the influence of an onshore wind. The oil was carried up and deposited on the beach by wind and wave action as the tide receded. The oil remaining on the water surface after completion of the full tidal cycle was collected with skimmers. Booms were deployed to prevent cross-contamination between test and reference areas and were left in place for several weeks to contain oil sheen (Sergy and Blackall 1987). The dispersed oil release was achieved using a subsea diffuser pipe placed on the bottom just outside the study area. The currents provided the mixing and movement patterns required to distribute the oil throughout the study area (Sergy and Blackall 1987).

> A pycnocline is a boundary layer separating two layers of different densities. A large density difference between surface waters and deep ocean water prevents vertical mixing. Formation of pycnoclines may result from changes in salinity or temperature. Because the pycnocline zone is stable, it acts as a barrier for surface processes.

The untreated surface slick was stranded in the intertidal zone in a manner representative of an accidental spill of moderate severity (Sergy and Blackall 1987). Relatively stable conditions were reached after about 48 hours, when oil on the beach was no longer being refloated in large quantities. During this 48-hour period, about one-third of the oil was recovered from the water surface by skimming, one-third remained stranded on the beach and one-third was evaporated or dissolved in sea water (Sergy and Blackall 1987). Over the next two years, the original amount of stranded oil was reduced by about 70%. Sergy and Blackall (1987) commented that this rate of natural shoreline cleaning was surprisingly high, given the short period during which it could occur and the protected nature of the shoreline. Biodegradation was considered to have removed only minor quantities of stranded oil; the majority of oil removal was attributed to physical processes. Oil residues were highly visible on the beach after two years; however, the distribution of oil was very patchy. The majority of the oil remaining on the beach was incorporated into an asphalt pavement. Some of the oil from the beach was transported by runoff and wave action to the adjacent subtidal sediments; however concentrations were still relatively low after two years (Boehm et al. 1987). *In situ* biodegradation could not be confirmed; therefore it was considered a negligible factor in the removal of sedimented subtidal oil at the low concentrations encountered (Sergy and Blackall 1987).

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Concentrations of oil in the water column were lo w beneath the initial slick and only trace amounts were measured over the subsequent two years.

The subsea discharge of oil/seawater/dispersant was swept by currents through the designated study areas. Oil contacted bottom sediments and organisms from the shoreline to a depth of 15 m. Within a few days the oil was distributed throughout the surface waters of Ragged Channel, including the reference site. The subtidal benthos received average exposures of 300 mg/kg, 30 mg/kg and 3 mg/kg in the two study Bays and the reference Bay, respectively (Boehm et al. 1987). Concentrations of oil in the water column were diluted to near background levels within a period of days. There was minimal recoalescence of oil on the water surface and negligible oiling of intertidal sediments.

Conditions in the immediate vicinity of the subsea release caused benthic organisms to be exposed to high concentrations of toxic aromatic hydrocarbons, which would otherwise have been rapidly lost to surface evaporation (Sergy and Blackall 1 987). The dispersed oil was introduced at greater water depths than would likely be the case of a surface oil slick. Sergy and Blackall (1987) noted that in many Arctic coastal areas, the often present and well -pronounced pycnocline would suppress the down ward mixing of oil to productive benthic depth zones. The high, short-term exposures to the chemically-dispersed oil produced acute behavioural and physiological effects in a wide variety of animals and, in a few species, a short - term reduction in abundanc e (Sergy and Blackall 1987). Responses included the emergence from the sediment and/or immobilization of infaunal and epibenthic invertebrates. Recovery and reburial occurred in the two weeks following the spills. No behavioural changes were observed at th e other study sites, which were exposed to very low levels of dispersed oil.

Heterotrophic bacteria in the water column were not affected by the surface oil slick and only temporarily affected by the chemically-dispersed oil. Effects on bacteria living in subtidal sediments were also minor (Bunch 1987).

There was no major large-scale mortality of benthic infauna to either oil release. After the two-year post-spill monitoring period, there was no evidence of large -scale mortality of subtidal benthic biota attributable to either the chemically-dispersed oil or the oil-contaminated beach. There were few changes in the populations or community structure of infauna, epifauna or macroalgae (Cross and Thomson 1987 ; Cross et al. 1987a ; Cross et al. 1987b). There were no measured biological effects in the intertidal zone due to the absence of Arctic intertidal life.

There were indications that exposure to the persistent oil residues in subtidal sediments (from both spills) was responsible for medium term (1 -2 year) sublethal effects, but only in a few species. For example, there were effects on condition in the bivalve *Macoma calcaerea* and decreases in density of the polychaete *Spio* spp. (Cross and Martin 1983).

Exposure to even low concentrations of chemically-dispersed oil resulted in a rapid and significant uptake by subtidal benthic fauna, particularly the filter -feeding bivalves (Humphrey et al. 1987 a). Bioaccumulation was observed over a relatively large area due to the extensive vertical and horizontal movement of the oil cloud. The water column was only a short-term source of oil exposure. Body burdens of filter -feeding bivalves were reduced within two weeks and those of all organisms were reduced considerably within the first year. On a longer term but very localized basis, deposit -feeding benthos living in contaminated sediments had elevated body burdens two years post -spill. In some cases an uptake-depuration balance appeared to exist.

Indirect effects of bioaccumulation were not part of the scope of the BIOS Project. Sergy and Blackall (1987) suggested that tainting may occur in shellfish harvested for human consumption. They also suggested that bioaccumulation may be of ecological significance in concentrated feeding areas for birds and mammals that use benthic biota as a food source. They stated that critical habitat for resource species

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(e.g. fish, birds and mammals) should be protected from oil of any type, and particularly from floa ting and beached oil slicks.

8.2.2 The shoreline study

The shoreline study evaluated options for cleanup of oiled shorelines by comparing cleanup methods to natural self-cleaning. Small plots (20-40 m^2) were used in the intertidal and backshore zones of the test beaches (Sergy and Blackall 1987). The same Lagomedio crude was used as for the near-shore study. Oil was deposited mechanically. Both emulsified and non-emulsified oil were used in paired plots (**Figure 8.3**). After application of oil, the plots were left for 24 hours prior to initiation of cleanup tests. This was assumed to represent minimal response time.

Figure 8.3. Shoreline study test plots during the BIOS project. *Image courtesy of Ed Owens.*

Wave energy was the dominant factor in the removal of oil from exposed beaches. On the partially exposed beach, the majority of oil was removed within the first open water season and by the end of the second season less than 0.03% remained.

Rising tides removed large quantities of oil on sheltered beaches over the first two days, after which relatively stable conditions prevailed. Oil was less persistent in the fine-grained sediments than on the pebble-cobble beach, where oil residues were still visible four years post-spill (Sergy and Blackall 1987). Climate and soil conditions played major roles in the fate of oil in the backshore plots. Significant amounts of oil remained in surface and subsurface sediments after three years; however there was considerable change in the composition of the oil due to weathering and biodegradation (Sergy and Blackall 1987).

Cleanup techniques evaluated were based on the operational realities of the eastern Canadian Arctic (Lee et al. 2011). Small labour force and the impracticability of disposing of large volumes of contaminated materials were the primary limiting factors; therefore, the emphasis was placed on the selection of techniques that would either have low labour or simple waste disposal requirements. The methods chosen were: *in situ* combustion; mechanical mixing of contaminated sediments; application of chemical surfactants to disperse stranded oil (BP1100X and Corexit 7664); and application of a solidifying agent to the stranded oil.

Chemical solidification was effective in stabilizing the oil but very labour intensive. Low-pressure flushing by sea water of oiled fine-grained beach sediments did not reduce oil concentrations. Burning also proved ineffective as even high-temperature igniters failed to sustain combustion on oiled cobble beaches. Application of commercial fertilizers to the plots increased bacterial numbers and degradation of oil on fine-grained backshore sediments but not on those of coarse material (Eimhjellen and Josefson 1984). Based on the experimental design (i.e., pseudoreplication), methodologies available (e.g., lack of genomic analysis to study whole microbial community responses) and the type/amount of data collected

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that limited statistical rigour, Sergy and Blackall (1987) could only suggest that biodegradation, enhanced or natural, was unlikely to be a factor of quantifiable significance except over a long period of time.

Wave energy was insufficient to mix surfactant and oil on very sheltered beaches; therefore, this treatment was ineffective in these areas. The use of surfactants quickly reduced the amount of oil present on the surface of cobble beaches partially exposed to waves. Sergy and Blackall (1987) concluded that surfactants would be most effective on small sections of coast where stranded oil might otherwise have a severe short-term impact or where long-term persistence was not desirable. Furthermore, consideration of the effects of flushing the oil from the beach into adjacent near-shore waters would be required.

Mechanical mixing of oiled intertidal and backshore areas reduced the total hydrocarbon concentrations in surface sediments, but in many cases at the expense of increasing subsurface concentrations (Sergy and Blackall 1987). The technique could be applied where the objective is to reduce contamination of surface traffic, to prevent or reverse asphalt pavement formation, or, in the backshore, to increase rates of weathering and to enhance biodegradation.

8.2.3 Important factors affecting the consequences of the experimental BIOS spills

8.2.3.1 Oil properties and behaviour

- One-third of the near-shore undispersed oil spill evaporated.
- Wave action was the primary mechanism for removal of oil from beaches.
- The majority of the residual oil on the beaches was incorporated into asphalt pavements.

8.2.3.2 Effect of the environment on fate and behaviour

- Rising tides removed large quantities of oil on sheltered beaches over the first two days, after which relatively stable conditions prevailed.
- Oil was less persistent in the fine-grained sediments than on the pebble-cobble beach, where oil residues were still visible four years post-spill.
- Biodegradation was considered a minor factor, but the evidence was inconclusive, especially in light of findings from recent studies using 'genomic' techniques.
- Oil persisted for long periods of time on low-energy beaches and backshore areas and in nearby seabed sediments.
- Climate and soil conditions played major roles in the fate of oil in the backshore plots.

8.2.3.3 Oil toxicity

- Biological impacts from both near-shore spills (dispersed and undispersed) were relatively minor and effects were short-term with no evidence of large-scale mortality of subtidal benthic biota and few changes in community structure of infauna, epifauna and macroalgae.
- Exposure to the persistent oil residues in subtidal sediments (from both dispersed and udispersed near-shore spills) was responsible for medium-term (1-2 year) sublethal effects, but only in a few species.
- The subsurface injection of dispersed oil caused benthic organisms to be exposed to high concentrations of toxic aromatic hydrocarbons that would otherwise have been rapidly lost to surface evaporation; responses included the emergence from the sediment and/or immobilization of infaunal and epibenthic invertebrates. Recovery and reburial occurred in the two weeks following the spills.

8.2.3.4 *Spill response*

- Skimming removed approximately one-third of the undispersed surface oil spill.
- Chemical solidification was effective in stabilizing the oil but very labour intensive.
- Low-pressure flushing by seawater of oiled fine-grained beach sediments did not reduce oil concentrations.
- Burning was ineffective as even high-temperature igniters failed to sustain combustion on oiled cobble beaches.
- Application of commercial fertilizers to the plots increased bacterial numbers and degradation of oil on fine-grained backshore sediments but not on those of coarse material.
- Wave energy was insufficient to mix surfactant and oil on very sheltered beaches; therefore, this treatment was ineffective in these areas.
- The use of surfactants quickly reduced the amount of oil present on the surface of cobble beaches partially exposed to waves.
- Mechanical mixing of oiled intertidal and backshore areas reduced the total hydrocarbon concentrations in surface sediments, but in many cases at the expense of increasing subsurface concentrations.

8.2.4 *Lessons learned from the BIOS project*

- According to Sergy and Blackall (1987), the BIOS Project provided no major ecological reasons to prohibit the use of chemical dispersants on oil slicks in near-shore areas similar to the experimental sites. Despite the high exposure of benthic organisms to chemically-dispersed oil, there were no major population- or community-level consequences. However, for use in areas where benthic organisms (such as filter-feeding bivalves) are consumed by wildlife or humans, the authors raised concerns over bioaccumulation.
- Sergy and Blackall (1987) stated that chemical dispersion may be the only alternative in situations where the immediate protection of shoreline and near-shore habitats is of primary importance or where a shoreline cleanup operation is environmentally less desirable. They further suggested that where practical and effective application methods and dispersant formulations are available, it would seem appropriate to give pre-spill approval for dispersant use along sections of Arctic coastline with ecosystems typified by the Cape Hatt site.
- Exposure to low-level oil residues in the sediments did not cause significant changes in the subtidal benthic populations over a two-year period following the spill.
- Oil residues persisted for long periods of time on low-energy beaches and backshore areas and in nearby seabed sediments; therefore, cleanup decisions should consider the implications of long-term oil residues to shoreline users and the potential for chronic bioaccumulation or sublethal effects in subtidal benthos.
- The results confirmed that Arctic beaches can be cleaned of oil by natural processes, despite the short open water season, and that this can occur very quickly on beaches exposed to moderate wave action.
- Sergy and Blackall (1987) stated that the BIOS findings show that natural cleaning of oil-contaminated beaches can be an environmentally acceptable option for low-priority shorelines similar to those at Cape Hatt. Response efforts would be more effectively directed toward areas of greater importance and sensitivity, such as shores adjacent to communities, wildlife breeding and staging areas and traditional hunting and fishing camp locations.
- Of the cleanup methods studied, two—surfactant flushing and mechanical mixing—produced an immediate reduction in the quantity of oil on the beach surface. These might be desirable where the objective is to reduce contact between oil and wildlife that frequent the shoreline.

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- Washing wit h surfactant could also be considered as a means to prevent oil -sediment consolidation (asphalt pavement) in the intertidal zone.

8.2.5 Research needs based on the BIOS project

<u>Recommendation:</u> The fate and behaviour of oil (dispersed and undispersed) released due to a subsea blowout in the Arctic requires much more study. Models to predict the trajectory of dispersed oil in open water and under ice are needed. The models will require real -time high resolution meta-data of oceanographic properties to provide emergency response support (Lee et al. 2011).

<u>Recommendation:</u> The relative risks of bioaccumulation across trophic levels in Arctic subtidal communities should be assessed and the results of the assessment used to set research priorities. Risks should be esti mated for organisms which consume subtidal benthic species, as well as the benthic species themselves.

<u>Recommendation:</u> The comparative risks of un dispersed and dispersed oil spills under Arctic conditions to fish, sea -birds and marine mammals should be ass essed. The risk scenarios should include subsea use of dispersants.

<u>Recommendation:</u> Future experimental spill in any environment must use statistically rigorous procedures involving randomized and replicated plots.

8.3 Other Arctic Oil Spill Field Trials

8.3.1 Review of conclusions arising from experimental spills in ice covered waters

Dickins (2011) reviewed research into the behaviour of oil spills in ice covered waters. The Dickins review is comprehensive and provides a base upon which to develop research programs in the Canadian Arctic. At the time of the review, Dickins, noted that the SINTEF Oil in Ice Joint Industry Partnership Program conducted between 2006 and 2009 in the Norwegian Barents Sea, was the most comprehensive Arctic oil spill research program completed. The field trials component of this program was between 2008 and 2009. The SINTEF projects have investigated: oil distribution and bioavailability; fate and behaviour of oil spills in ice; *in situ* burning (ISB) of oil spills in ice; mechanical recovery of oil spills in ice; use of dispersants; and chemical herders on oil spills in ice. Field trials have demonstrated that oil spilled in ice conditions was significantly thicker than in open water, and the final area of spilled oil was much wider in open water (Lee et al. 2011). Thus, oil contained in ice and snow would be thicker and more easily burned or otherwise recovered. Furthermore, containment in ice reduce s wave action , and relatively slow weathering in the Arctic environment can provide time for spill response, extendin g the window of opportunity for mobilizing response activities and enhancing the effectiveness of certain response measures such as burning and skimming (Dickins 2011).

Lee et al (2011) provided a summary of conclusions drawn from experimental oil spills in the Arctic:

- Low water and air temperatures in additi on to ice -covered waters generally result in greater oil equilibrium thickness due to smaller contaminated areas and reduced spreading rates;
- Ignitability is enhanced in cold temperatures and ice due to the greater persistence of lighter and more volatile components of petroleum hydrocarbons;
- Ice has a dampening effect on waves; therefore, the sea conditions in areas of the Arctic may be significantly less severe than most open ocean areas, thus allowing for easier marine operations;
- Ice can naturally contain spilled oil and act as a barrier to spreading. The natural herding properties of oil then enhance the effectiveness of ISB by thickening the slick;

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- High ice concentrations (7/10 or more) tend to immobilize and encapsulate most spilled oil quite rapidly, particularly from a subsea blowout;
- Ice encapsulated oil is effectively isolated from weathering and allows for a longer window of opportunity to carry out cleanup activities. Effective combustion at a later date is possible because the oil remains unweathered; and
- Most of the Arctic shoreline has a seasonal fringe of fast ice that acts as an effective natural barrier against oil contamination on the coastline in winter.

Lee et al. (2011) also provide d a summary of some of the challenges an d limitations associated with response to oil spills in the Arctic:

- Oil trapped on or under ice in moving pack ice is difficult to measure because crews cannot maintain continuous operations with immediate means of evacuation;
- Skimming operations can be hindered d ue to the slow spreading and flow of oil in leads and openings in pack ice;
- Moving vessels during response operations can cause rapid spreading of oil, creating a thinner, less easily recoverable or ignitable slick; and
- Crude oils with pour points of 0 °C or less tend to gel.

8.3.2 Research needs regarding spills in Arctic conditions

According to Dickins (2011), future advances in the ability to respond to spills in ice will require a new approach to permitting experimental spills in the Canadian Arctic. Lee et al. (2011) stated that Arctic field trials could be used to address research questions, such as:

- Remote sensing systems to detect, monitor and map the transport and spreading of oil in ice and below ice;
- Weathering of oil in cold water conditions;
- Development and verification of oil-in-ice drift and fate models;
- Improved mechanical response, such as skimmers, pumping systems for viscous oil and the removal of oil from ice and water;
- ISB in broken ice;
- Oil dispersion enhancement by dispersants and enhanced OMA formation;
- Bioremediation of oil stranded on shorelines;
- Characterization of water-soluble components and biological effects on Arctic species; and
- Development of operational endpoints for spill cleanup operations.

Two major initiatives have been recently funded to address some of these oil spill response knowledge gaps in the Arctic:

The Arctic Oil Spill Response Technology Joint Industry Programme (JIP) was initiated in 2012 and is currently ongoing. It represents a collaboration of ten international oil and gas companies to enhance industry knowledge and capabilities in the area of Arctic spill response, as well as to increase understanding of potential impacts of oil on the Arctic marine environment under the management of the International Associat ion of Oil and Gas Producers. The JIP (described at: www.arcticresponsetechnology.org) has several specific projects each focusing on a different key area of oil spill response:

- Project 1 - Fate of Dispersed Oil under Ice: The project will provide important information for dispersants use in ice -covered marine environments and develop a tool to support contingency planning.

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- Project 2 - Dispersant Testing under Realistic Conditions: The project will define the operational criteria for use of dispersant and mineral fines in Arctic marine waters with respect to oil type, oil viscosity, ice cover (type and concentration), air temperatures and mixing energy (natural, water jet and propeller wash). Another objective is to identify the regulatory requirements and permitting process for dispersant and mineral fines use for each Arctic nation/region.
- Project 3 - Environmental Impacts from Arctic Oil Spills and Oil Spill Response Technologies: The project will improve the knowledge base for using "Net Environmental Benefit Analysis" (NEBA) for response decision-making and ultimately facilitate stakeholder acceptance of the role of EIA in oil spill response plans and operations.
- Project 4 - Oil Spill Trajectory Modeling in Ice: The project will advance the oil spill modeling for oil spills in ice -affected water s by evaluating ice trajectory modeling approaches and integrating the results into established industry oil spill trajectory models.
- Project 5 - Oil Spill Detection and Mapping in Low Visibility and Ice: The project will expand remote sensing and monitori ng capabilities in darkness and low visibility, in pack ice and under ice. This project is split into two elements: surface remote sensing (i.e. satellite -borne, airborne, ship-borne and on -ice detection technologies) and subsea remote sensing (i.e. mobile -ROV or AUV based and fixed detection technologies).
- Project 6 - Mechanical Recovery of Oil in Ice: The project will evaluate novel ideas for improving efficiency of mechanical recovery equipment in Arctic conditions.
- Project 7 - *In Situ* Burning of Oil in Ice-Affected Waters. State of Knowledge: The project aims to raise the awareness of industry, regulators and external stakeholders to the significant body of knowledge that currently exists ISB.
- Project 8 - Aerial Ignition Systems for *In Situ* Burning: The project will develop improved ignition systems to facilitate the use of ISB in offshore Arctic environments, including ice when the presence of sea ice restricts use of vessels as a platform for this response option.
- Project 9 - Chemical Herders and *In Situ* Burning: The project will advance the knowledge of chemical herder fate, effects and performance to expand the operational utility of ISB in open water and in ice-affected waters.
- Project 10 – Field Research: Results from previous research projects show that many of the advances in our state of knowledge about Arctic response technology were gained through controlled field experiments with oil. This project will pursue opportunities for large scale field releases for validation of response technologies and strategies.

Within Canada, the Churchill Marine Observatory was established in July 2015, with research infrastructure funding is provided through the Canada Foundation for Innovation's Innovation Fund, Aboriginal Affairs and Northern Development, the Province of Manitoba and collaboration with the universities of Calgary and Victoria. This facility will be a dedicated multi -disciplinary research facility for study on the impact of oil spills in sea ice, as well as the investigation issues facing arctic marine transportation.

<u>Recommendation:</u> The effects of climate change on ice cover and ice behaviour in the Canadian Arctic is an important issue requiring investigation.

<u>Recommendation:</u> Explicit research questions regarding effects on marine biota require further study and validation both for spills under ice-covered conditions and in open-water conditions.

<u>Recommendation:</u> Effects of cleanup activities on Arctic biota require study in order to support comprehensive and reliable NEBA.

<u>Recommendation:</u> If future multi-disciplinary field trials are allowed in the Canadian Arctic, they should be collaborative and incorporate the concerns and knowledge of Indigenous peoples.

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8.4 The Exxon Valdez Oil Spill (EVOS)

On March 24, 1989, the *Exxon Valdez* altered its course from shipping lanes to avoid floating ice and struck Bligh Reef in Alaska's Prince William Sound. The grounding was caused by a series of human errors committed by its crew members, including the master; however broader safety issues associated with inadequate policies and procedures within the Exxon Shipping Company, as well as the U.S. Coast Guard were identified by the NTSB (1990). Specific policy and procedure issues included: failure of the Exxon Shipping Company to provide a fit master and a rested and sufficient crew; the lack of an effective Vessel Traffic Service; inadequate personnel training; deficient management oversight; and the lack of effective pilotage services (NTSB 1990).

The remote location, accessible only by helicopter, plane or boat, made government and industry response efforts difficult, as did severe weather with high winds, which came about two and half days later. About 42,000 tonnes of Prudhoe Bay crude oil (Alaska North Slope medium crude oil; Table 2.2) was released into the ocean affecting an area of about 28,000 km^2. The dispersant Corexit 9580 was applied on the day of the spill, but there was not enough wave action to mix the dispersant with the oil in the water (Chapter 6). Some of the surface oil was burned, reducing 113 m^3 of surface oil to a removable residue; however, unfavourable weather prevented further burning. Booms and skimmers (Chapter 6) were deployed for mechanical recovery, but skimmers were not readily available during the first 24 hours, and thick oil and kelp tended to clog the equipment. A major storm 2.5 days after the spill led to extensive mousse formation. About 20-30% of the spilled oil evaporated, was diluted or dispersed into the water column or photooxidized (Chapter 2) (Wolfe et al. 1994). Within three weeks, less than 25% of the oil remained on the sea surface.

Immediate effects of the spill included large-scale seabird mortality (estimates vary but average about 250,000 comprising over 90 species), as well as the deaths of about 1,000 to 5,500 sea otters (Platt and Ford 1996; Ballachey et al. 2014). Twenty-two killer whales died. One pod of highly-exposed killer whales lost seven members within a week of the spill, including three adult females, and an additional seven or eight members of this pod died over the next two years (Spies et al. 1996). Other species affected directly by mortality included river otters, harbour seals and bald eagles, plus unknown numbers of herring, salmon and other fish species (Ballachey et al. 2014).

About half of the oil was distributed along the shoreline and inter- and subtidal areas as far as 970 km from the spill site (Owens 1991; Wolfe et al. 1994)). About 782 km of Prince William Sound (about 16% of shoreline) and 1,315 km of Gulf of Alaska (about 14% of shoreline) were oiled to some degree (Owens 1991; Neff et al. 1995).

Shoreline cleanup included manual removal and the use of surfactants and high-pressure hot water (Chapter 6). The cleanup focused on the intertidal zone, including removal of tar mats, mousse and subsurface deposits, as well as sediment relocation and biostimulation to enhance microbial degradation. An assessment in 1990 showed that most remaining deposits were isolated from the biological environment (with the exception of exposure pathways leading from mussels to sea otter (Chapter 4); therefore, only a few larger deposits of subsurface oil were relocated to the middle and upper intertidal zones where it was subject to wave and tidal action and natural cleaning. Less intrusive cleanup efforts continued through the summer of 1991 involving mostly manual removal and bioremediation. Although biostimulation on an unprecedented scale (48,600 kg N in various nutrient formulations applied at 1,400 different sites) initially accelerated oil removal from shoreline surfaces (Atlas and Hazen 2011), after a few years even the deliberately untreated ('set-aside' or control) sites were equivalent in terms of residual oil load (Pritchard et al. 1992).

The aggressive cleanup methods had significant effects. For example, high-pressure washing (**Figure 8.4**) removed all of the macroalgal and mussel communities in some areas, extending the time to recovery of

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intertidal communities (State of Alaska 1993). Furthermore, the high -pressure washing, while removing oil from the upper and mid-intertidal zone where its effects were somewhat restricted to relatively tolerant organisms, such as barnacles, rockweed and mussels, transported the remobilized oil into the lower intertidal and shallow subtidal zones where the oil was placed into contact with relatively more sensitive organisms, such as hard shelled clams and crustaceans (State of Alaska 1993). The physical features of the shoreline were also affected; e.g., silty sediments were washed out of th e beach areas and into the water. Most of the animals that normally lived in these beach areas required a certain mix of fine -grained sediments; therefore, many would not return until the beach sediments had stabilized (Shigenaka 2014).

Figure 8.4 Cleanup workers spray oil -covered rocks on Prince William Sound with high -pressure hoses. *(Photo Courtesy of Exxon Valdez Spill Trustee Council)*

Eight years after the spill, removal of oil was attempted on armoured portions of beaches in Prince William Sound using high -pressure injection of a surfactant (Brodersen et al. 1997). The cleaning was conducted at the request of the local subsistence-use community. Cleaning resulted in a mean reduction of 62% at treated areas with further reduction occurring in the following year. However, newly visibly oiled sites were exposed by winter storms the year after the cleanup.

Shigenaka (2014) stated that despite the unprecedented scale, duration and cost of the response, modeling of the fate of the spilled oil estimat ed that the cleanup itself removed only a small portion (a little more than 10%) of the spilled oil from the environment. By far, the largest part of the total was naturally weathered or degraded (**Figure 8.5**).

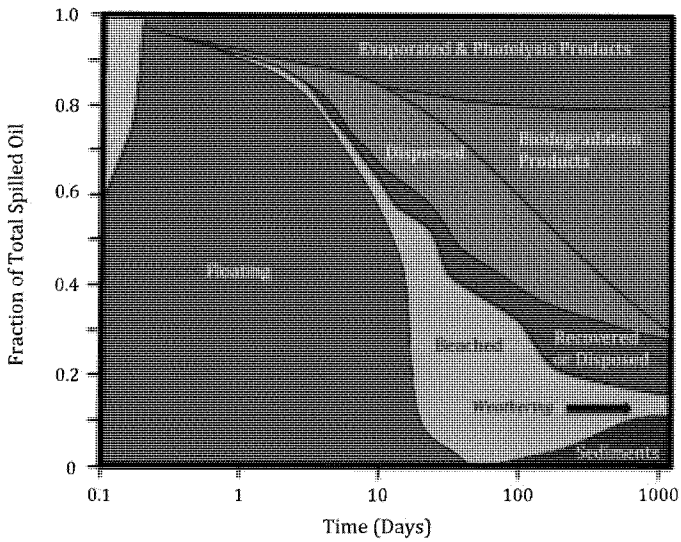

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Figure 8.5. Modeled fate of the spilled Exxon Valdez oil with time. *The portion recovered by the cleanup is in green (after 1,000 days). Reprinted with permission from Wolfe, D.A., Hameedi, M.J., Galt, J.A., Watabaytashi, G., Short, J., O'Clair, C., Rice, S., Michel, J., Payne, J.R., Braddock, J., Hanna, S. and D. Sale. 1994. The fate of the oil spilled from the Exxon Valdez. Environmental Science and Technology 28: 561–568. Copyright (1994) American Chemical Society.*

The *Exxon Valdez* Oil Spill Trustee Council (EVOSTC) has adopted an official list of resources and services injured by the spill as part of its Restoration Plan. This list includes fish and wildlife resources that "experienced population-level or chronic injury from the spill" (EVOSTC 2014). The list is divided into six categories (**Table 8.1**). The EVOSTC Restoration Plan was first adopted in 1994 when it was created as guidance for the expenditure of public funds.

Table 8.1 Exxon Valdez Oil Spill Trustee Council List of Resources and Services injured by the Spill

Recovering	Recovered	Not Recovering	Recovery Unknown	Human Services
• Designated wilderness areas • Intertidal communities • Killer whales – AB Pod • Sediments	• Archaeological resources • Bald eagles • Barrow's goldeneyes • Clams • Common loons • Common Murres • Cormorants • Dolly Varden • Harbour seals • Harlequin ducks • Mussels • Ink salmon • River otter • Sea otters • Sockeye salmon	• Marbled murrelets • Pacific herring • Pigeon Guillemots • Killer whales – AT1 population	• Kittitz's murrelets	• Commercial fishing • Passive use • Recreation and tourism • Subsistence

Some authors maintained that the persistent, residual oil in unconsolidated sediments of the intertidal zone continues to contaminate invertebrates to the extent that some vertebrate consumers (including fish, otters and seabirds) are still being exposed to toxic oil concentrations (Peterson et al. 2003; Ballachey et al. 2014). In contrast, other authors have concluded that the Prince William Sound ecosystem has effectively recovered from the EVOS (Harwell and Gentile 2006).

There appears to be a consensus that continued contamination of subtidal areas is not a concern (Ballachey et al. 2014). The authors noted that some of the estimated 55,000 kg of oil remaining in mid- and upper-intertidal habitats may persist at some sites for several decades and that some of the residual oil is largely unweathered because it exists in anaerobic patches where biodegradation is slow (Figure 2.3; Chapter 3). This estimate is for subsurface oil in the upper two-thirds of the intertidal zone. Inclusion of the lower one-third and surface oil would substantially increase this estimate (Short et al. 2004).

Potential chronic effects to the pink salmon population have been linked to direct embryo mortality in the first five years after the spill and continuing exposure to residual oil. Ballachey et al. (2014) summarized the results of several field and laboratory studies of pink salmon, and concluded that these studies "provided compelling evidence of chronic impacts to pink salmon from oil persisting in intertidal

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habitats". They cited evidence including: 1) observations of elevated embryo mortality in the five years following the spill; 2) identification of the exposure mechanism from contaminated beaches to spawning gravels; and 3) measured effects on fitness following embryonic exposures under laboratory conditions. They concluded that the group of pink salmon studies they reviewed provided "unprecedented evidence that exposure of embryonic life stage (*sic)* to low level PAHs (in ppb) from persistent oil can have a population level effect". Ballachey et al. (2014) also concluded that chronic exposure to oil and possibly latent effects of acute exposure appeared to have decreased survival and constrained recovery of the sea otter population in Prince William Sound for more than two decades.

Not all authors agree that population-level effects on pink salmon persisted. In a detailed review of studies of effects on pink salmon in Prince William Sound, Brannon et al. (2012) concluded that problems with experimental design and methods called into question the potential for population-level effects. Brannon et al. (2012) stated that there was no evidence that weathered oil increased toxicity, either in the form of total PAH or as HPAH (high molecular weight PAH) leachate. They went on to state that field evidence suggested that "well-weathered oil deposits in the actual areas impacted by the EVOS do not represent a continuing and increasing threat of interstitial toxic water to pink salmon eggs incubating in adjacent streams". Further, the authors examined studies of growth rate of juvenile pink salmon in Prince William Sound and concluded that the assumptions necessary to discern growth effects either were not met or contained uncertainties about their validity. As discussed in Chapter 4, the tributaries to Prince William Sound produce so many juvenile fish that any deficit in production in tributaries affected by the spill did not appear to affect total abundance.

The effect of spatial scale may underlie much of the debate regarding effects on pink salmon. Hatchery production of pink salmon in Prince William Sound is comparable in numbers with wild stock production. Hatchery production was not affected by the EVOS, and effects on wild stock production were negligible at the scale of Prince William Sound as a whole. However, pink salmon spawning streams that bisected beaches that were heavily oiled by the EVOS were subject to mortality and sublethal effects on embryos, as stated by Ballachey et al. (2014).

Herring have been the focus of extensive studies since the EVOS because they were commercially harvested in Prince William Sound and their numbers showed large declines within a few years of the spill. The herring population was still very low in 2014 with no commercial fishery present; however, the role of the spill in the decline and lack of recovery of herring, relative to other factors, including disease, predation and recruitment, is unclear and, according to Ballachey et al. (2014), is unlikely that it will ever be completely understood.

> *"If the Exxon Valdez experience has taught us anything, it has emphasized the importance of variability as both a key feature of biological communities and a critical consideration to integrate into assessment of disturbance and recovery. As we inevitably consider oil spill scenarios for the Arctic, they are framed against the background of change that is occurring at unprecedented rates"* (Shigenaka 2014).

Peterson et al. (2003) emphasized that there can be delayed population reductions and cascades of indirect effects that postpone recovery. For example, harlequin ducks appeared to be affected via the energetic costs of metabolizing PAHs, leading to lower body mass and elevated overwintering mortality. According to a study by Iverson and Esler (2010 cited by Ballachey et al. 2014), full recovery of the harlequin duck population would require from 16-32 years under best-case and worst-case scenarios, respectively. Peterson et al. (2003) pointed out that there were also cascade effects from the cleanup actions. For example, the removal of *Fucus* stands with their associated community of grazers, such as limpets and periwinkles, led to initial blooms of green algae and opportunistic barnacle growth with declines in invertebrates associated with the *Fucus* canopy. After regrowth of the *Fucus*, there was

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another mass mortality in 1994, probably caused by simultaneous senility of a single -aged stand. All of this extended the recovery process for a decade or more.

The Shigenaka (2014) analysis concluded that the intertidal biota of Prince William Sound had recovered, at least in terms of the metrics used in the analysis (attainment of parallel temporal trends in abundance) , after an initial six -year period. This conclusion applied to infauna, algae and invertebrates living on the surface of rocky substrates. The authors emphasized that their conclusions were for intertidal populations only.

The *Exxon Valdez* Oil Spill Trustee Council (2015), Harwell and Gentile (2006) and Shigenaka (2015) all note that while the spill and cle anup clearly caused significant ecological effects for months to a few years post-spill, *natural variability and the occurrence of multiple anthropogenic stressors not associated with the spill are now making the detection of any potential residual effects of the spill very difficult.* In addition, current natural and human stressors may be hindering recovery of some resources initially injured by the spill. The passage of time and the evolution of science have shifted the purpose and utility of the Injured Resources and Species List. "The Council recognizes that the complexities and the difficulties in measuring the continuing impacts from the spill result in some inherent uncertainty in defining the status of a resource or service through a specific list an d the Council's focus has accordingly expanded to a more ecosystem approach" (EVOSTC 2015).

The inherent natural variability of biological communities complicated interpretation of monitoring data following the spill (Shigenaka 2014). A new statistical app roach to analysis was designed , which acknowledged that: unoiled reference sites may be biologically different from oil ed sites at the time of impact, thus rendering absolute convergence of conditions in the recovery phase less useful; and conditions at both oiled and unoiled locations are likely influenced by a host of factors not related to the spill disturbance (Shigenaka 2014). The new approach analysed long -term datasets for patterns of abundance. The underlying assumption was that in the absence of sp ill effects, sites would respond similarly to climate, ocean conditions or other determinants of biological communities in a defined study region.

The risk of oil spills must be assessed against the backdrop of responses to large -scale phenomena. The Shigenaka (2014) analysis noted that results across three independent experimental studies revealed a strong correlation of biological metrics in Prince William Sound with cycles of the Pacific Decadal Oscillation (PDO). For example, mussels and molluscs appear to respond positively to warmer cycles of the PDO whereas rockweed shows greater abundance during cool phases. The author stated that in recent years, linkages between changes in Alaskan biological communities and conditions and large -scale atmospheric, oceanic and climatic shifts have grown more numerous.

The use or interpretation of data in support of specific values or policies is called 'normative science', and, according to Landis (2007), normative science can explain at least some of the differences in interpretation of data related to long -term effects of the EVOS. According to Landis, ecosystem health, ecosystem integrity, ecological significance and re covery are constructs that incorporate values and policies. "Separation of science from policy or at a minimum a transparent acknowledgement of the science-policy interaction is clearly necessary to obtain a clear picture of the ecological system under investigation" (Landis 2007). Landis suggested two alternatives for dealing with normative science: 1) to stop using terms such as ecological significance, integrity and recovery (in the Clementsian context) [3]; or 2) to understand that ecological policy is a c omplex and multi -component decision-making process that cannot be summarized in metaphorical clichés , such as ecosystem health . If a policy goal is being discussed or supported, it should be clearly defined and obvious to the reader. Landis (2007) asked

[3] *Clementsian ecology is based on assumptions of steady-state equilibrium and development of predictable and specific climax communities, which have fallen out of favour given the complex and dynamic properties of ecosystems.*

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whether controversies such as those surrounding long -term effects of the EVOS are really debates about policy rather than scientific merit.

8.4.1 Important factors affecting the consequences of the Exxon Valdez spill

8.4.1.1 Oil properties and behaviour

- Evaporation occurred at lower rates due to low temperature.
- Natural dispersion was wind driven and very widespread.
- Stranding was widespread on shorelines.
- Remobilization of oil deposited on shorelines occurred by tidal and wave action.

8.4.1.2 Effect of the environment on fate and behaviour

- Biodegradation was slow but significant at the ambient temperatures present in Prince William Sound.
- Oil persisted in unconsolidated sediments of the intertidal zone.
- Some persistent oil remained in a relatively unweathered state due to low oxygen conditions in sediments.
- Wave exposure enhanced the weathering of non-sequestered residual oil.

8.4.1.3 Oil toxicity

- Acute toxic effects of floating oil from the spill inclu ded large -scale seabird and sea otter mortality.
- Chronic toxicity caused mortality -related damage to the social structure of a killer whale pod . Indirect effects may have postponed recovery of some species (e.g., energetic costs of metabolizing PAHs caused lower body weight and higher winter mortalities in harlequin ducks) . Long-term effects of residual oil on pink salmon were caused by exposure of early life-stages and may have been limited to sub-populations using spawning streams that bisected beaches that were heavily oiled.
- Intertidal biological communities recovered relatively rapidly from the combination of oil toxicity and the effects of cleanup activities , although there was a mass mortality of *Fucus* in 1994 (associated with simultaneous senility of a single -aged stand), which extended recovery of areas affected by the mortality by a decade or more.

8.4.1.4 Spill response

- Limited regional response capability and the remoteness of the location with no ground access delayed the response.
- Severe weather following the spill limited some response options , such as ISB and use of dispersants, so that the spread of the surface oil was increased and more oil was entrained into intertidal sediments on affected shorelines.
- Aggressive shoreline cleanup methods, such as high temperature and pressure washing , impaired site recovery rates due to mortality and removal of indigenous species and habitat destruction.

8.4.2 Lessons learned from the Exxon Valdez spill

- Challenges inherent in oil spill response in northern, remote areas prone to adverse weather increase the probability that o il released from near -shore tanker accidents will rapidly disperse and reach shorelines over a wide area.

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- Some response measures can be difficult or impossible to implement because of weather and sea conditions (e.g., the use of dispersants or ISB).
- Aggressive shoreline cleanup methods , such as high -pressure washing , can impart substantial negative effects on biological communities, including indirect and cascade effects, and can delay recovery.
- Cleanup removed a small proportion of the spilled oil (about 10%).
- Weathering and biodegradation were of primary importance with respect to the nat ure and extent of residual oil.
- Altering the physical features of a beach or shoreline can significantly affect the recovery of impacted plants or animals. Physical recove ry and stabilization of a site are necessary for biological recovery.
- Long-term, population -level effects are difficult to study and the results can be controversial, which emphasizes the critical need for baseline data on natural resources , as well as con sensus-based definitions of monitoring endpoints.
- The inherently high degree of natural variability found in systems such as Prince William Sound can limit or preclude the use of standard or traditional statistical methods. Risks of oil spills in the Arctic must be framed against the background of large-scale oceanic and climatic phenomena and changes that are occurring at unprecedented rates (Shigenaka 2014).
- NOAA found that so-called 'set-aside sites' that were oiled but intentionally left uncleaned have been critical to the NOAA monitoring program's ability to determine impacts due to oiling alone and those due to cleanup, and to enable scientifically rigorous interpretation of data.
- Normative science may have played a role in the debate regarding long -term, population -level effects of the spill. This is not surprising given the high social, economic and cultural value of the receiving environment. The consequences of being wrong about the interpretation of data can be ecological and societal . Therefore, both consequences should be identified and considered in a transparent manner.

8.4.3 Research needs related to the Exxon Valdez spill, in the Canadian context

<u>Recommendation:</u> The most commonly -used suites of biological metrics for Canadian receiving environments (e.g. intertidal or subtidal community metrics) should be examined across broad gradients of natural factors and over longer time -frames to establish the relative roles of local, regional and global factors. Research should be conducted on a range of spec ies and/or communities with various levels of resilience and should focus on identified high risk marine environments (note: WSP and SL Ross, 2014a,b).

The relative role of local, regional and global factors in determining the natural variability of biological metrics is poorly understood, particularly at the population and community level. Factors, such as climate and oceanographic variability, overexploitation and invasive species , may overwhelm any signal of subtle, long-term effects of residual oi ls. However, the incremental stress produced by exposure to oil may contribute to 'tipping points' for species whose resilience has already been challenged by other stressors. Furthermore, future field studies should incorporate multiple study areas (both reference and treated) to enable determination of experimental error, which might mitigate the effect of natural variability affecting scientific conclusions.

8.5 Deepwater Horizon (DWH) Blowout

The DWH offshore oil drilling rig was situated in the Macondo oil prospect in the Mississippi Canyon, a valley in the continental shelf of the Gulf of Mexico. The well over which it was positioned was 1,522 m below the surface and extended 5,486 m into the geological formation under the seabed. On the night of April 20, 2010, a surge of natural gas blasted through a concrete core recently installed to seal the well.

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The natural gas travelled up the rig's riser to the platform where it ignited, killing 11 workers and injuring 17. The rig capsized and sank on the morning of April 22, rupturing the riser, through which drilling mud had been injected to counteract the upward pressure of oil and natural gas. Without any opposing force, Macondo light crude oil (Table 2.2) began to discharge into the Gulf (Britannica Online 2015).

Several unsuccessful attempts were made to slow or stop the flow of oil, a technically challenging task given the extreme depth of the blowout. A cap was successfully placed in June that greatly reduced the flow, but didn't eliminate it. The well was permanently sealed with the successful completion of a relief well on September 19, 2010. The spill is considered the largest accidental marine oil spill in the history of the petroleum industry. The estimated volume released varied from about 430,000 to 500 ,000 tonnes (McNutt et al. 2012; Fingas 2013 ; Chapter 6). According to satellite images, the spill directly affected 180,000 km^2 of ocean (**Figure 8.6**). Response measures included over 1,300 km of containment booms to either corral the surface oil or function as barriers to protect coastal areas, shrimp/crab/oyster ranches or other ecologically sensitive areas. About 7,000 m^3 of two different Corexit chemical dispersants were used, including about 2,920 m^3 injected into the subsurface oil plume at the wellhead during the spill (Chapter 6). Oil was removed from the water surface via ISB, in addition to physical recovery (e.g., booming and skimming).

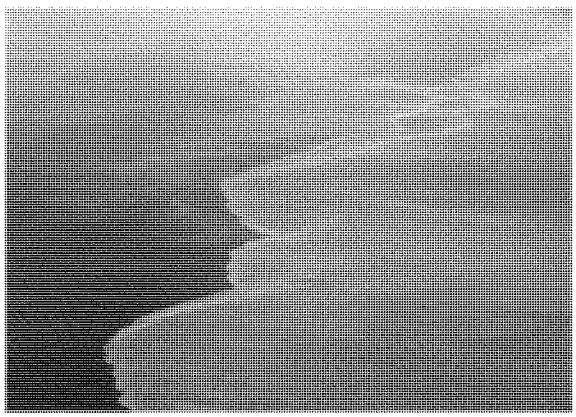

Figure 8.6 Oil from the Deepwater Horizon oil spill approaches the coast of Mobile, Alabama, May 6, 2010. *Image from US Navy.*

According to the Federal Interagency Solutions Group (2010), the expected mass balance was: 17% of the oil was collected as it was released from the well; 5% was burned; and 3% was skimmed from the surface for a total of 25% (**Figure 8.7**). The fate of the 75% of the oil not collected, burned or skimmed was estimated as follows:

- 13% dispersed naturally;
- 16% dispersed with Corexit;
- 23% evaporated or dissolved; and
- 23% other mechanisms such as stranded on shorelines or sinking to the seafloor.

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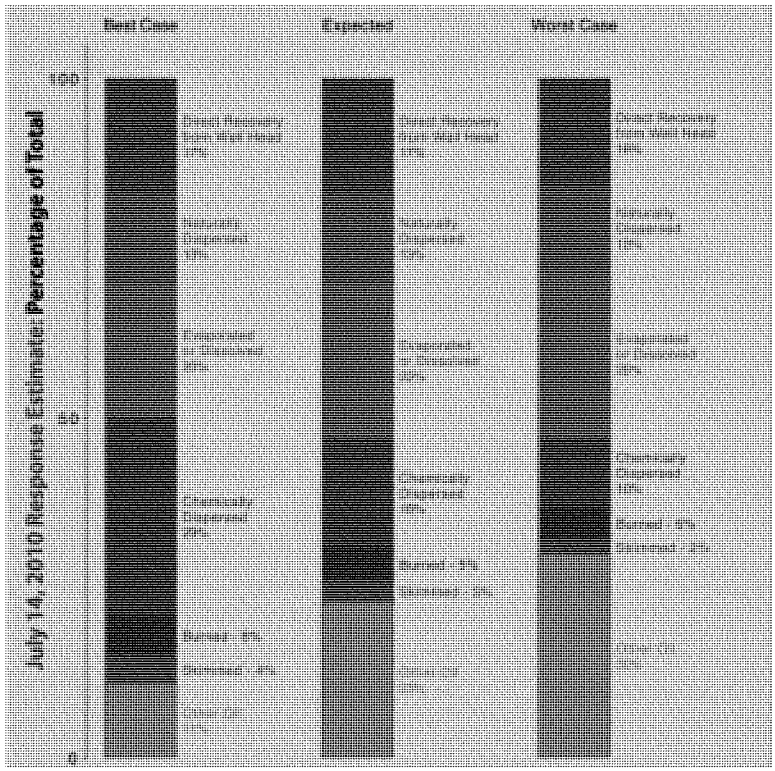

Figure 8. 7. Oil budget estimates expressed as percentages of the cumulative volume of oil discharged through July 14, 2010. *These estimates served as a guide for the response to the Deepwater Horizon blowout. Image from Federal Interagency Solutions Group (2010).*

A more recent review of the mass balance concluded that of the total amount of oil released, about 6 to 7% of the oil was burned *in situ*, about 12 to 15% of the oil was skimmed, and about 6 to 26% (best estimate about 10%) of the oil arrived onshore (Fingas 2013). Fingas estimated that 50 to 55% of the oi l remained in the water column.

Subsequent results have shown that sinking played a more important role than previously thought. Valentine et al. (2014) suggest t hat about 318,000 m^3 of submerged oil from the subsurface plume may have been trapped in deep ocean intrusion layers at depths of about 1,000 -1,300 m. Based on spatial, chemical, oceanographic and mass balance considerations, they calculated that between 4 -31% of oil sequestered in the deep ocean was deposited in patches to deep sea sediments. The pattern of contamination points to deep ocean intrusion layers as the source and was most consistent with dual modes of deposition: 1) a 'bathtub ring' formed from an oil-rich layer of water impinging laterally upon the continental slope (at a depth of ~900–1,300 m); and 2) a higher-flux 'fallout plume' where suspended oil particles sank to underlying sediment (at a depth of ~1,300–1,700 m). Independently, Chanton et al. (2015) also calculated that a large proportion of the 'missing' Macondo oil has been partially buried in deep sea sediments in the Gulf of Mexico. The precision of these estimates is still debated due to sample heterogeneity. Furthermore, natural oil seeps, with a fingerprint similar to the Maconda oil, also contributed to uncertainty within the oil budget. In terms of shoreline impact assessments, it was difficult to distinguish tar mats or tar balls originating from the spill from those originating from seeps.

8.5.1 Dispersant use

During the spill response, approximately 6.9 million litres of Corexit were applied , 4 million litres of Corexit 9527 and 9500A at the surface and 2.9 million litres of Corexit 9500 via subsurface injection (Federal Interagency Solutions Group 2010).

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One of the controversial aspects arising from the DWH oil spill response operation was the injection of dispersants at the wellhead located at 1,500 m depth. Monitoring programs identified the presence of a plume of oil at a depth between 1,100 and 1,200 m below the surface (Camilli et al. 2010; Reddy et al. 2011). While the plume was initially thought to be direct evidence of dispersant effectiveness, a number of studies suggested that the subsurface dispersants had little impact in dispersing the spill or preventing oil from reaching the surface (e.g., Paris et al. 2012; Peterson et al. 2012). They argued that the turbulence in the jet of gas and oil from the well head was sufficient to induce massive dispersion. Both processes likely occurred. In addition to changes in oil droplet size (Johansen et al. 2013; Brandvik et al. 2013), evidence supporting the benefits of the subsurface dispersant application included aerial photographs taken during the DWH oil spill that showed a loss and gain in the magnitude of the surface slick following subsurface dispersant injection and its shutdown.

The trade-offs regarding the use of dispersants are the subject of debate since the dispersants may also enhance the bioavailability of the spilled oil or be toxic on their own. In a recent review of the pros and cons of dispersants, Prince (2015) concluded that in most cases, the potential environmental costs of dispersant use are likely outweighed by the much shorter residence time of dispersed oil in the environment. Notwithstanding this conclusion, the debate continues.

The fate and effects of the dispersants with associated dispersed oil within the water column of the Gulf of Mexico is the subject of considerable uncertainty. Kujawinski et al. (2011) found that the concentration of dioctyl-sodium sulfosuccinate or DOSS (a key ingredient of these dispersants) was sequestered in deepwater hydrocarbon plumes at 1,000-1,200 m water depth and did not intermingle with surface dispersant applications. They also found that the concentration distribution was consistent with conservative transport and dilution at depth and it persisted up to 300 km from the well, 64 days after deepwater dispersant applications ceased. Thus, the surfactant does not appear to have been rapidly biodegraded. This finding agrees well with the laboratory results reported by Campo et al. (2013), who showed that at 5 °C DOSS degraded slowly by only 61% in triplicate microcosms by day 42.

While Corexit 9500 and Corexit 9527 are ranked within the low to moderate toxicity level for aquatic species (George-Ares and Clark 2000), a level of sensitivity remains over their application. Evidence from studies with invertebrates and fish have shown that oil dispersed as small droplets in the water column was more bioavailable, and therefore more toxic, than the oil alone (Chapter 4). However, while the results of laboratory studies have raised much media attention over the high risks of dispersant use, ecological relevance must be considered. Following its dilution in an 'open sea' environment, it is unlikely that the dispersants would be present in concentrations toxic to pelagic organisms (Kujawinski et al. 2011; Lee et al. 2013). Nevertheless, caution must still be taken as most toxicological studies have been based on short exposure periods (e.g. 96 h LC50 tests with standard 'regulatory' reference test organisms). Many of the long-term Natural Resource Damage Assessment (NRDA) studies on the long-term effects of oil and chemically-dispersed oil on trophic level dynamics in the Gulf of Mexico have not yet been released, and no community/population level studies have been conducted on chronic effects of the chemically-dispersed oil on deepwater organisms.

Microbial communities in the Gulf of Mexico rapidly responded to the spill, including in deepwater plumes (Atlas and Hazen 2011). Redmond and Valentine (2012) showed that the deepwater microbial community was very different from the surface microbial community, reflecting the colder temperatures at the plume depth, as well as exposure to natural oil seeps in the Gulf and possible leakage from other oil wells in the area. The authors suggested that the high natural gas content of the spill may have provided an advantage to specific taxa, notably *Colwellia* and to a lesser extent *Oceanospirillales* (Chapter 3).

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8.5.2 *Deposition and effects on shorelines*

A review by Michel et al. (2013) of the extent and degree of shoreline oiling stated that about 1,770 km of shoreline was affected consisting of 51% beaches, 45% marshes and 4% other shoreline types. Simulations of oil transport from the footprint of the spill on the water surface showed that the mass of oil that reached the shorelines was between 9,000 and 27,200 tonnes, with an expected value of 19,960 tonnes (Boufadel et al. 2014). It should be noted that this predicted mass is much lower than estimated by Fingas (2013). Shoreline cleanup was authorized on 550 km of shoreline. Two years after the spill, oil remained on 687 km of the shoreline but to a much lower degree (e.g., the authors reported that the heavily oiled category declined from 360 km to 6.4 km using the Shoreline Cleanup and Assessment Technique (SCAT)). The bulk of the oil stranded during a three-month period when many of the beaches were in an erosional state that led to burial of the oil. In addition, oil was stranded high in the supratidal zone due to high water levels and wave activity. The oil was buried, exposed and remobilized multiple times in some areas. Removal of deeply buried oil required extensive mechanical and manual excavation and sieving. In the lowest intertidal/near-shore subtidal zones, some of the oil/sand mixture accumulated in the near-shore subtidal zone forming extensive submerged oil residue mats. Elsewhere, the oil/sand residues adhered to relict[4] marsh platforms composed of clay and peat at the toe of sand beaches —these mats that were exposed only during the lowest of tides and/or buried by beach accretion were difficult to remove. In marshes, the oil tended to strand along the marsh edge and spread no more than about 10-15 m inland. Some of the most heavily oiled marshes were cleaned using: intensive manual and mechanical raking and cutting to remove the oiled vegetation mats and wrack (Chapter 6); careful removal or reduction of the thick oil layers on the substrate; and limited application of loose organic sorbents.

Microbial communities on oiled shorelines showed a distinct response to the contamination, both in terms of overall bacterial numbers and on the abundance of known oil degraders. Kostka et al. (2011) showed that bacteria in Pensacola Beach sands were on average two to four orders of magnitude more abundant in the presence of oil contamination, and high cultivatable counts as well as nucleic acid-based analyses supported the premise that the majority of bacteria in the oiled sands were active. The authors noted that the native microbial communities responded fairly quickly to oil contamination and suggested that conditions in subtropical sands (temperature, oxygen supply and nutrients) appear to favour a broader diversity of hydrocarbon degraders that might render biostimulation (via nutrient addition) unnecessary.

The impacts of the oil spill on shoreline vegetation were variable. Along some heavily oiled shorelines, there was nearly complete flora mortality. Moderate oiling had no significant effect on some species (e.g. *Spartina;* cord-grass) but significantly lowered live above-ground biomass and stem density of others (*Juncus;* rushes) (Mendelssohn et al. 2012). Since the spill, some recovery has been noted (Mendelssohn et al. 2012). However, there were concerns about whether some shorelines would revegetate naturally before shoreline erosion occurred. Biber et al. (2014) found that there was more rapid removal or degradation of oil along coastlines in Mississippi exposed to higher energy levels. Plant recovery was more rapid in these locations. In contrast, at low-energy locations, oil was still detected in sediments and on plants one year post-spill and plants in these locations exhibited chronic stress, which depressed photosynthesis. Silliman et al. (2012) reported that while rapid salt marsh recovery was observed, there were also permanent marsh area losses. They also observed thresholds of oil coverage that were associated with severity of salt marsh damage; plant death of marsh edges more than doubled rates of shoreline erosion, further driving marsh platform loss that is likely to be permanent. However, in non-eroded areas, marsh grasses had largely recovered. The authors noted that heavy oil coverage on shorelines that were already experiencing elevated erosion because of intense human activities induced a geomorphic feedback that amplified erosion and thus limited the recovery of otherwise resilient vegetation.

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[4] *A group of animals, plants or objects that exists as a remnant of a formerly widely distributed group in an environment different from that in which it originated.*

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8.5.3 *Effects on fisheries*

Near-shore fisheries were vulnerable to the spill both in the spawning grounds in the Gulf of Mexico and in estuarine nursery areas. The spill overlapped with peak spawning periods for several important species, including brown shrimp (*Farfantepenaeus aztecus*), white shrimp (*Litopenaeus setiferus*), blue crab (*Callinectes sapidus*) and spotted seatrout (*Cynoscion nebulosus*) (Mendelssohn et al. 2012). Although the location of the spill was in deep water, currents carried oil into the shallow spawning areas of these species. The short- and long-term effects of this oil (and/or dispersants) on eggs and larvae are still uncertain, but Fodrie and Heck (2011 cited by Mendelssohn et al. 2012) did not find short-term negative effects on juvenile fish associated with inshore seagrass beds. Some evidence of low impact to fish populations was noted at sites distant from the heavily oiled Louisiana coast (Anderson 2014). Although coastal fishes likely have adapted for shifting habitat availability in Louisiana, it is uncertain whether wetland losses due to the spill would have a negative effect on fish production (Anderson 2014). Fishery responses to the spill are difficult to tease out because of high annual variability and the effects of many other anthropogenic stressors (Mendelssohn et al. 2012). Anderson (2014) emphasized the importance of continued monitoring of sediments, plants and animals in oiled areas.

A study of the effects of the spill on spotted seatrout by Brown-Peterson et al. (2014) showed a substantial, but short-term effect on reproductive parameters after the spill compared with historical, pre-spill data. The authors noted that the availability of pre-spill (baseline) reproductive data allowed direct comparisons from the same sites before and after the spill. Important environmental variables, such as temperature and salinity, were similar during most of the months of the reproductive season pre- and post-spill, and the two significant differences in temperature and salinity pre- and post-spill were deemed to be unimportant biologically. The authors acknowledged that other impacts on the spotted seatrout populations in the years between the two sampling events, such as heavy fishing pressure, habitat loss and changes in salinity due to drought and flood conditions, may also contribute to differences in reproductive parameters, "although impacts from DWH seem to be the most parsimonious explanation for the observed differences" (Brown-Peterson et al. 2014). Uncertainty regarding the interpretation of results is increased because measured PAH concentrations in water taken from the sampling sites in 2011 were generally below the 1.03 μg/L detection limit and were not significantly different from pre-spill concentrations. The authors suggest that larval spotted seatrout may have been exposed to oil during the summer of 2010, leading to altered reproductive dynamics in 2011 as the fish reached sexual maturity. Reproductive parameters were returning to pre-spill levels by July, but the effective reproductive season was reduced from the normal six months to three months.

Remedial operations may have caused significant secondary effects. The creation of a 35-km oil prevention berm off the Chandeleur Islands, a unique habitat for neonatal and juvenile lemon sharks (*Negaprion brevirostris*), may have reduced available habitat for the sharks and may also have inhibited sharks pupped on the windward side of the island from reaching the protected sea grass beds on the leeward side (McKenzie et al. 2014). A lack of suitable protected nursery habitat may have, in turn, resulted in a further reduction of lemon shark numbers around the islands. The authors noted that it will be important to continue monitoring the population numbers and the occurrence patterns to determine if the effects are long-term.

8.5.4 *Effects on invertebrates*

Studies on the impacts of the DWH oil spill on blue crab (Fulford et al. 2014) and oysters (Le Peyre et al. 2014) revealed that effects, if any, were indirect (for blue crab) or the data indicated no significant differences in biomarkers of exposure to PAHs (for oysters). Fulford et al. (2014) found that megalopal (final crab larval stage) settlement patterns were more likely a reflection of climatic conditions and pointed out that cyclic uncertainty in recruitment of decades or longer is important and should be accounted for in interpreting data time series. They also found a low level of larval mortality due to PAH

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exposure in 2010 in the wild, which was supported by laboratory assays that showed a mortality effec t only at PAH concentrations of 1 mg/L or higher. They suggested that direct larval exposure to oil in the pelagic zone may not be the most important point of vulnerability for the crab population . Rather, effects might be indirect, operating at the larval source and, in turn, affecting larval delivery to the near -shore habitat.

The nature of the spill in terms of its magnitude and release at depth raised concerns regarding effects on deepwater coral communities. White et al. (2012) studied coral communitie s at 11 sites three to four months after the well was capped. Healthy coral communities were observed at all sites greater than 20 km from the well. However, at one site 11 km southwest of the well, coral colonies showed several signs of stress. Of the corals examined at the affected site, 46% exhibited evidence of impact on more than half of the colony, and nearly a quarter showed impacts to more than 90% of the colony. Floc deposited on the corals was traced to the Macondo well via analysis of hopanoid bi omarkers (White et al. 2012) . The significance of particulate deposition (i.e., mortality caused by smothering) from the failed 'top kill' operation and/or the episodic release o il (with similar biomarker ratios) from natural seepage (which has been reported to occur within the region) has not been fully addressed.

8.5.5 Effects on trophic level dynamics

There is currently much interest within the NRDA process to identify potential community and population level changes, as well as alterations in trophic level dynamics. Graham et al. (2010) found that $\delta^{13}C$ depletion, used as a tracer of oil -derived carbon in mesozooplankton and suspended particulate samples in near -surface and bottom waters from four stations in 8 to 33 m water depth in the northern Gulf of Mexico, corresponded with the arrival of surface slicks from the DWH oil spill and demonstrated that carbon from the spill was incorporated into both trophic levels of the planktonic food web.

Results of *in situ* microcosm experiments performed in North Inlet Estuary, SC, with DWH and a Texas crude oil showed a decrease in chlorophyll *a* in phytoplankton as crude oil concentration increased from 10 to 100 µL/L (Gilde and Pinckney 2012). This change was interpreted as a decrease in biomass rather than chlorophyll due to the short duration of the experiments. Diatom, cyanobacteria, euglenophyte and chlorophyte abundances were unaffected or increased with increased oiling, whereas cryptophyte abundance decreased. The a uthors suggested that oiling could result in changes in phytoplankton community composition within salt marsh estuaries impacted by oil from the DWH, thereby affecting higher trophic levels, such as zooplankton, which selectively feed on phytoplankton that might be killed by the oil (Gilde and Pinckney 2012).

Tarnecki and Patterson (2015) recently reported a potential shift in the diet (prey including fish, decapods, cephalopods, stomatopods, gastropods, zooplankton and other invertebrates) and trophic posi tion (determined from stable isotope ratio-mass spectrometry analysis of $\delta^{13}C$, $\delta^{15}N$, and $\delta^{34}S$) for red snapper (*Lutjanus campechanus*) from the north -central Gulf of Mexico following the DWH oil spill. Stable isotope data indicated a post -spill increase in red snapper trophic position (^{15}N enrichment) and an increase in benthic versus pelagic prey (^{34}S depletion), that was consistent with observed dietary shifts, likely linked to relative abundance of prey resources.

Indirect effects on the menhaden (*Brevoortia tyrannus*) population in the Gulf occurred due to trophic cascade effects (Jeffrey Short, pers.comm . 2015). The effects began with the death of close to 1 million seabirds due to physical contact with the oil. The mortality of birds increased the abunda nce of juvenile menhaden—a primary food source for the birds. The menhaden population increased to unprecedented abundance and biomass levels in 2011 -2012. H owever, this was accompanied by a decline in the condition of the fish (and thus their oil content) as they exhausted their food supply. These effects occurred on a much larger spatial and temporal scale than direct toxicity from the spill. As of 2015, the menhaden population appeared to be recovering.

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8.5.6 *Effects on ecosystem services*

An ecosystem services approach can supplement traditional methods of assessing or valuing damage to natural resources by estimating flows of goods and services before and after an event (NRC 2013; reviewed in Chapter 4). The approach focuses not on the natural resources themselves, but on the goods and services these resources supply to people.

> The ecosystem services approach provides a useful framework regarding how to manage human activities (and the consequences of those activities) in order to sustain the ecological structure and function necessary to provide essential services. The approach can be used to inform the public and decision-makers about the connections between human activities and the effects of those activities on ecological services. However, in order to advance the approach beyond generalizations, data are required which describe explicit connections between ecosystems and benefits (economic, cultural and spiritual). These data are fragmentary and scattered among many disciplines.

Ecosystems are subject to natural disturbances, such as floods, droughts and disease outbreaks, as well as human-caused disturbances, including oil spills. Ecosystems are also subject to slowly changing long-term stresses, such as nutrient enrichment and changes in the sediment supply, as observed in the Gulf of Mexico. These long-term stresses can affect the ability of the system to respond to a shock, such as the DWH spill (NRC 2013).

Ecosystem resilience to shocks from oil spills in the context of long-term stresses is key to the maintenance of ecosystem services. In some cases, systems will undergo fundamental shifts in structure and function following disturbances (e.g., changes in trophic structure caused by invasive species in the Great Lakes). Although resilience to preserve ecosystem services is an important concept, the understanding of complex and highly variable systems, such as the Gulf of Mexico (or the Gulf of St. Lawrence) is insufficient to support specific, resilience-based recommendations regarding response to or monitoring of oil spills. If policies and decisions are to be based upon the ecosystem services concept, a large, coordinated, long-term, multi-disciplinary effort will be required.

NRC (2013) summarized the effects of the DWH spill on four key components of the Gulf of Mexico that provide ecosystem services. Their conclusions are presented below.

Wetlands

Acute effects on marshes, where the biota are not expected to recover, appear to be confined to the edges of bays, canals and creeks in a limited subset of the oiled wetlands.

- Where the vegetation has died and root systems have been lost in heavily oiled areas, the erosion of sediment is leading to the conversion of once-productive marshland to open water;
- Subsequent tropical storm activity resulted in additional erosion of oiled marshes; and
- Based on numerous studies that document a rapid recovery from oiling and a relatively low sensitivity of perennial marsh vegetation to hydrocarbons, marsh vegetation can be expected to suffer little or no long-term impairment in areas where roots and rhizomes survived the initial impact of oil fouling. If roots and rhizomes do not survive, then an area will likely not recover on its own due to the loss of habitat by erosion.

These impacts need to be viewed in the context of significant and continuing losses of wetlands in the Gulf of Mexico due to many other stressors, including subsidence, canal dredging, salt intrusion and sediment starvation.

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Fisheries

Despite long-term studies and ongoing development of models, the ability to detect spatial and temporal differences in fishery productivity in the Gulf of Mexico is limited. Recent developments in fishery data collection, such as the introduction of vessel monitoring systems in the reef fish fishery, could improve estimates of abundance. However, any mortality or reduction in individual fitness caused by the spill directly or indirectly may take years or, for some species, decades to transfer through the ecosystem and be observed.

The direct value of commercial fisheries to the fisherman is calculated using the dockside value of the catch minus any expenses incurred to capture those fish. The method for evaluating the economic effects of an oil spill on commercial fisheries is derived from either reduced production (due to mortality or fishery closures) or by reduced consumer demand (due to the perception of reduced fish quality or safety).

The immediate economic impact of the DWH spill was a 20% decrease in commercial fish catch for 2010.

Marine Mammals

Bottlenose dolphins were the representative marine mammal examined by the NAS because of the role dolphins play in three ecosystem services—regulating, supporting and cultural. As apex predators[5], dolphin populations serve as important indicators. They are the most studied and among the most popular and charismatic marine mammals. The stranding of hundreds of dolphins in the Gulf of Mexico before, during, and after the spill stimulated considerable public concern. NRC (2013) suggested that if the post-spill mortality event is determined to be linked to the spill, then an opportunity may exist to establish a plan to protect and restore the dolphin habitat and to reduce dolphin mortality due to human activities. Venn-Watson et al. (2015) studied unusual mortality events of bottlenose dolphins (*Tursiops truncates*) in Louisiana, Mississippi and Alabama from 2010 to 2014 and concluded that the rare, life-threatening and chronic adrenal gland and lung diseases identified in stranded dolphins were consistent with exposure to petroleum compounds. The authors suggested that the DWH spill was a contributor to increased dolphin deaths.

Deep Gulf of Mexico

The deep sea area of the Gulf is so vast and sampling is so sparse that gaps in knowledge inhibit the ability to apply an ecosystem services approach in a quantitative way. The NRC (2013) assumed that the primary ecosystem services of the deep Gulf are supporting, e.g., resupply of nutrients. In addition, the NRC (2013) stated that the deep sea area of the Gulf provides the regulatory service of pollution attenuation, e.g. via bacterial degradation. The release of crude oil from the Macondo well at depth created a unique opportunity to study deep sea oil biodegradation.

8.5.7 Important factors affecting the consequences of the Deepwater Horizon Blowout

The huge volume of spilled oil and the length of time it took to stop the release were primary factors in determining consequences. The presence of significant anthropogenic stressors prior to the DWH spill, including degraded or lost shoreline habitats, overexploitation of some fishery resources, salt intrusion, sediment starvation and nutrient influxes leading to algal blooms and oxygen depletion, added confounding factors to the interpretation of the effects of the spill. In addition, the Gulf of Mexico is a highly complex ecosystem subject to major natural cycles.

[5] *An apex predator is a predator residing at the top of a food chain on which no other creatures prey upon.*

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8.5.7.1 Oil properties and behaviour

- Dissolution of BTEX and naphthalenes into the water column occurred as the oil moved to the surface from the well.
- There was rapid evaporation of lighter fractions in the floating oil.
- Substantial spreading of surface oil occurred driven by wind and currents.
- Biodegradation along shorelines and within dispersed oil was significant, with a fairly quick response of native microbial degraders to the oil.
- Sinking of physically- and chemically-dispersed oil droplets from mid-column occurred.

8.5.7.2 Effect of the environment on fate and behaviour

- High microbial degradation rates occurred along shorelines and in shallower waters of the Gulf because of subtropical temperatures (Chapter 3).
- Rapid penetration of oil occurred along sand shorelines due to high porosity.
- Repeated burial and exposure of oil occurred (and continues to occur) along shorelines.
- Wave action led to stranding of oil in the high intertidal zone.
- Heavy coating of vegetation along the margins of wetlands and estuarine shorelines including the formation of 'oiled' subsurface mats.

8.5.7.3 Oil toxicity

- Mass acute mortality occurred due to exposure to floating oil.
- Death of vegetation occurred along heavily oiled margins of marshes.
- The extent of chronic toxicity is difficult to determine given multiple confounding factors.
- There may have been trophic level effects, including a shift in the diet of red snapper and a temporary increase in menhaden abundance originating with the mass mortality of predatory seabirds.
- Chemical and biological markers of oil exposure and toxicity were not often included in case studies of oil effects on aquatic biota.
- Extrapolation of laboratory and mesocosm study results to the Gulf of Mexico ecosystem is very difficult.

8.5.7.4 Spill response

- Novel application of dispersants in the deep subsurface was associated with the formation of a deepwater plume.
- Surface application of dispersants had limited effect in reducing oil reaching shorelines.
- Deployment of oil booms, berms and other barriers protected specific shoreline or island areas.
- ISB removed 6-7% of the oil.

8.5.8 Lessons learned from the Deepwater Horizon Blowout

8.5.8.1 Lessons learned according to the USGS (2015)

- Oil was consumed by bacteria, seafood was not contaminated by hydrocarbons or dispersants and the oil budget was by and large accurate. There was consensus that most of the oil was biodegraded. The only part of the oil budget that was later found to be inaccurate was the fraction of oil that was chemically -dispersed versus naturally -dispersed. That information had no impact on public safety, seafood safety or the response effort, but understanding the amount of oil that was dispersed chemically versus naturally is important for future such efforts.

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- The scale and complexity of DWH taxed the agencies involved.
- Future oil spill response preparedness should include the following actions:
    - Gathering adequate environmental baselines for all regions at risk;
    - Developing new technologies for rapid precise reconnaissance and sampling to support a timely and robust response effort;
    - Filling large information gaps regarding biological effects of oil, changing climate and other simultaneous drivers of variability in coastal and aquatic ecosystems;
    - Requiring that future oil extraction permits be conditional on having mechanisms in place to rapidly assess flow rate; and
    - Conducting research on the impacts of dispersants and dispersants -plus-oil on a wide range of species and life stages.
- The scale of the spill required unprecedented collaboration among government, academic and industry scientists and engineers. Scientific and engineering information was crucial to guide decision-making for questions never before encountered.
- The lack of peer -reviewed scientific publications from prior marine well blowouts was a significant drawback in addressing many of the issues.
- The event also showed the value of federal partnerships with academic institutions.

8.5.8.2 Other lessons learned (from the RSC Panel's review of the literature)

- Considering the size of the spill, relatively little oil made it to shore and tar balls and tar mats were distributed heterogeneously.
- Receiving environments that are already degraded by human activities (e.g. coastal marshes) have enhanced vulnerability to oil spills.
- Fate and transport of submerged, physically- and chemically-dispersed oil is poorly understood.
- The consequences of deposition of oil to deepwater sediments require investigation.
- The ability to determine effects of oil spills (even those as huge as the DWH) depends upon the availability of pre -spill baseline information, an understanding of natural variability, and an understanding of the effects of multiple anthropogenic stressors.
- Barriers to pr event oil incursion can have unintended consequences (e.g. , reduced available habitat for lemon sharks and inhibition of sharks from reaching the protected sea grass beds on the leeward side).
- Natural biodegradation on beaches in warm climates can be rapid and may not require nutrient supplementation, especially if available nutrients are already plentiful enough to support growth on the oil incursion. However, biodegradation will not remove all of the oil components.
- Effects on trophic dynamics via changes in biomass of primary producers and shifts in diet can occur. H owever, the spatial and temporal scales at which this could occur suggest that effects may not be widespread and/or long-term.
- Oil spill response plans should accommodate and exploit scientifi c opportunities and oil spill response should incorporate these opportunities.
- Experience of people involved in the response to the spill showed a pressing need for improved analytical procedures for the detection and characterization of oil spill constitu ents, including standard sample tracking and reporting procedures (Laboratory Information Management linked to sample collection) (Ken Lee, pers. comm.).
- Effects on ecosystem services can be very difficult to establish and quantify, particularly for resources where the ability to detect spatial and temporal differences is limited by lack of data or a lack of fundamental understanding of population or community structure, function and dynamics, or for ecosystem components that have had very little study (e.g, deep sea areas).

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- Even if effects on ecosystem services can be quantified, a comprehensive model is needed that incorporates biophysical, social and economic data for the Gulf of Mexico for the long -term (NRC 2013).

8.5.9 Research Needs Related to the Deepwater Horizon Blowout, in the Canadian Context

Recommendation: Research into the consequences of deposition of oil to deep sea sediments in the Arctic, as well as deep sea sediments south of the 60th parallel in Canada would provide part of the required knowledge base for support of decisions regarding offshore oil and gas exploration and oil transport. This is particularly important in light of frontier offshore oil and gas operations moving into deeper waters and emerging interest in the use of dispersants by subsea injection in the case of well blowouts. Offshore drilling in the Arctic and on Canada's east coast will create the potential for impacts on benthic ecosystems and commercial fisheries that have not yet been studied.

Recommendation: In the Canadian context, the current level of understanding of trophic dynamics in identified high -risk offshore and inshore marine habitats requires evaluation in order that critical gaps in understanding are identified.

8.6 Pine River, British Columbia Pipeline Break

On the night of August 1, 2000, a break in an oil pipeline released 952 m^3 of sour crude oil (BC -Light; Table 2.2) into the Pine River in northeastern British Columbia. The Pine River is a major tributary of the Peace River with mean flows in August of about 200 m^3/s (Water Survey of Canada 2015). About half of the oil entered the river and was dispersed downstream. The other half remained on land, contaminating surficial soils and groundwater. The Pine River is a source of drinking water for the town of Chetwynd.

The spill occurred in the vicinity of a previous spill of 30 m^3 of gasoline and 24 m^3 of diesel fuel on August 18, 1994 that caused the mortality of about 150 mature and 1,000 juvenile fish (Goldberg 2011).

8.6.1 Immediate effects, response and cleanup

As a result of the spill, Chetwynd closed its water intake and started exploring alternate water sources. A total of 1,637 dead fish were found 2-50 km downstream, comprised primarily of larger, more visible fish. Most were bottom-feeding species, consisting of 64% mountain whitefish (*Prosopium williamsoni*), 16% slimy sculpins (*Cottus cognatus*) and 5% burbot (*Lota lota*). The remaining fish were surface feeders, consisting of 5% bull trout (*Salvelinus confluentus*), 2% rainbow trout (*Oncorhynchus mykiss*) and 1% Arctic grayling (*Thymallus Arcticus*) (Alpine Environmental and EBA Engineering 2001). It was estimated that 15,000 – 27,900 fish mortalities occurred, based on numbers of fish per kilometer and assuming complete fish-kill in the first 30 km.

The recovery of oil from the river occurred over a period of about two months. According to Alpine Environmental and EBA Engineering (2001), about 91% of the oil was recovered or accounted for as follows:

- 447.5 m^3 entered the river;
- 358 m^3 recovered as liquid oil from the river and spill site prior to soil excavation;
- 89.5 m^3 lost to volatilization (calculated);
- 5 m^3 recovered from the river as liquid oil using absorbents;
- 21.4 m^3 recovered as liquid oil during soil excavation;
- 416 m^3 recovered from the spill site as part of soil excavation; and
- 83.2 m^3 not accounted for (included dissolved, adsorbed in soils along bank and in sediments, trapped in backwaters and eddies, trapped in logjams).

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A number of emergency response operations were deemed to have caused detrimental effects on fish and wildlife habitat. These included:

- The alteration of an 115 m-long backchannel from the break site to the river (to stop oil flow into the river) that resulted in the cut off of an oxbow channel that provided abundant fish food and the isolation of a back channel from fish access;
- Bank armouring to prevent erosion and the removal of logjams and riparian vegetation coated with oil removed floodplain habitat available to fish during high flows (Sumners 2001). In this case the contractor also failed to implement the consultant's recommended fish habitat features when installing the armouring;
- The removal of logjams and woody debris structures from the river channel without consultation and approval of regulatory agencies (Department of Fisheries and Oceans-Canada [DFO] and BC Ministry of Environment);
- Rerouting of the river near a tributary by the removal of a logjam diverted flow from a 3.4 km stretch of creek, the Lemoray Meander Loop. The new 1.6 km long channel that formed was unstable, having a steep gradient that resulted in mass erosion with vegetation, soils, gravels and sands washing downstream into spawning habitat—however there was no investigation to access the level of potential impact. While the logjam was eventually rebuilt, it was overwhelmed by the spring freshet in 2001 a 1-in2- or 1-in-3-year event). Sumners (2001) deemed this failure as "catastrophic to the stream channel and the habitat therein". Consulting engineers involved in the project concluded that the magnitude of the changes in the channel prohibited the effectiveness of further attempts at logjam construction; and
- Habitat damage was also caused by the actions of the regulatory agencies which allowed large machinery in the river channel because of the urgency to finish the cleanup work before winter.

Sumners (2001) noted that poor records were kept during the cleanup. In terms of remediating the damage to habitat caused by the initial emergency response operations, at least 13 logjams were rebuilt on advice from a consultant, primarily to maintain the stability of the river. Sumners (2001) expected that where jams were not rebuilt, others would likely form over time; however, the author concluded a net loss of habitat resulted.

8.6.2 Water quality

Visual and olfactory evidence of oil contaminants were evident over an 80-km section of the river following the spill. After three weeks, while concentrations of extractable petroleum hydrocarbons were still detected in waters downstream of the spill site, they were below water quality criteria (Alpine Environmental and EBA Engineering 2001), and few samples contained the more volatile fractions (Amec 2001a). By the end of August (one month later), petroleum hydrocarbons in the water had declined to below detection limits (Amec 2001a) and the presence of sheen was largely restricted to back eddies and other calm water areas along shorelines.

8.6.3 Sediments

Petroleum hydrocarbons accumulated in depositional environments within the river, in areas with soft, muddy sediments along the banks, in back eddies and other calm-water locations (Amec 2001b). There were also high concentrations of organic debris (e.g., branches, leaves and algae) that had accumulated along the shoreline, in front of logjams and attached to sweeper logs (overhanding trees with some limbs and branches submerged during high flows). Six PAH compounds and cadmium were detected in these materials at concentrations sufficient to be a concern for the health of fish and other aquatic biota (Amec 2001b). Heavy rainfall in late August and September mobilized previously stranded oil and oil-contaminated debris and sediments into relatively uncontaminated areas. Higher water levels also scoured some of the more heavily-contaminated depositional areas. The concentration of detectable oil

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concentrations declined by an average of 71% by October (Alpine Environmental and EBA Engineering 2001).

8.6.4 Periphyton and benthic invertebrates

Effects were observed on periphyton and benthic invertebrate communities immediately after the spill. A substantial increase in algal biomass that exceeded the apparent seasonal effects was observed at one sampling station (Alpine Environmental and EBA Engineering 2001). Total abundance of benthic invertebrates declined in August, with substantial recovery by November (Alpine Environmental and EBA Engineering 2001). The benthic invertebrate community structure was also affected. The effects on benthic invertebrates were of concern with respect to a decreased food supply for fish, especially young fish, during the recovery period between August and November—a critical period leading up to over-wintering (see discussion below).

8.6.5 Fish

Chemical analysis of sportfish sampled above and below the spill site did not indicate any trends in concentrations of total extractable hydrocarbons or PAH in liver and muscle tissue (Amec 2001c). Amec concluded that consumption of the Pine River fish caused no significant unacceptable risk to humans.

The initial spill of crude oil into the Pine River produced sufficient toxic material (especially that which dissolved to produce a water-soluble fraction likely a concentration of higher than 3 mg/L, Chapter 4) in the river that killed fish over a relatively short period of time (hours) (Birtwell 2003). Unfortunately, the actual concentrations in river water were not determined as samples were not collected at the time when the fresh crude oil was entering the river (Birtwell 2003).

Results of analysis of fish tissue and stomach contents indicated that longer-term risks to fish can be driven by feeding behaviour combined with oil dispersion and stranding on large woody debris. Analysis of fish bile showed no benzo(a)pyrene metabolites but some phenanthrene metabolites, indicating at least some PAH exposure (Alpine Environmental and EBA Engineering 2001). Although concentrations in whole fish were at levels of detection for total PAHs in surface feeders, predators and bottom feeders, the stomach contents of rainbow trout showed large quantities of unmetabolized PAHs. These fish were sampled at a logjam where a considerable amount of free oil was collected, suggesting that the fish were feeding within the pooled oil on drifting invertebrates. The hydrocarbons in the stomach contents mirrored the oil characterization results (Alpine Environmental and EBA Engineering 2001).

Exposure of fish to hydrocarbons can also be driven by temperature-related behaviour. Fish often conceal themselves in the substrate at temperatures less than 9 °C, particularly bull trout (Alpine Environmental and EBA Engineering 2001). If the concealment areas coincide with areas of oil deposition, the fish will be exposed. No consideration of hyporheic flows and effects on spawning gravels was apparent in the published reports about the spill.

The annual fidelity of fish to feeding areas could result in reduced food intake if prey species were less abundant or unavailable due to the impact of spilled oil (Birtwell 2003). Fish must obtain enough food prior to ice cover to survive the cold northern winter. However, if this is not accomplished and/or the metabolism of the fish is elevated due to exposure to contaminants and/or other stressful circumstances, survival is jeopardized (Lemly 1993 cited by Birtwell 2003). Because exposure to oil can elevate metabolism in fish, their survival under the colder winter conditions was a concern, especially at a time when metabolic activity and food intake usually decreases (Birtwell 2003).

The concerns about sublethal effects on fish led to an assessment of snorkel surveys conducted in the Pine River in 1993 (pre-spill), 1994 (after the 1994 spill), 2000 (about two months after the pipeline rupture), 2005, 2006 and 2007 (Goldberg 2011). The results suggested that the sportfish species composition

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(mountain whitefish, Arctic grayling, bull trout and rainbow trout) and abundance observed in 2005, 2006 and 2007 were similar to pre-spill observations (1993) in the river sections surveyed.

In summary, the 2000 spill caused mortality of several thousand fish, but there were insufficient studies to establish, with confidence, the nature and extent of sublethal effects on reproduction and recruitment. The concentrations of PAH compounds in fish stomach contents confirmed PAH exposure via the food chain. There was a potential for conc entrations of PAH in spawning habitat to have exceeded thresholds for sublethal effects, particularly for fall spawning species such as bull trout in 2000. However, there were no coordinated efforts to measure PAH concentrations and fish egg/embryo surviva l in spawning areas downstream of the spill.

8.6.6 Important factors affecting the consequences of the Pine River spill

8.6.6.1 Oil properties and behaviour

- Evaporation of light oil plus dis solution and natural dispersion were important factors leading to hydrocarbon c oncentrations within the water col umn of the river declining t o detection limits within one month.
- Physical interactions with shorelin es, sediments and woody debris led to substantial stranding of oil.

8.6.6.2 Effect of the environment on fate and behaviour

- Water flow led to rapid dispersion of acutely lethal concentrations of the residual oil up to 80 km downstream.
- Temperature declines in the fall triggered fish behaviour that increased exposure to oil in sediments.
- Substrate particle size was an important factor leading to retention of oil in fine sediments in slow-water areas.
- River channe l and shoreline characteristics were also important with respect to creating conditions that allowed oil to become associated with the river bottom (backwater and side channel areas).

8.6.6.3 Oil toxicity

- Substantial acute lethality to fish and benthic invertebrates occurred immediately after the spill.
    - Most fish deaths were of bottom-feeding species. Exposure may have been via both water and food organisms based on stomach content analysis, and
    - Mortality of benthic invertebrates was observed in the first month but rapid recovery was observed.
- There was possible increased overwintering mortality or increased susceptibility to predation (because of decreased fitness) caused by exposure to PAHs in overwintering habitats and possible effects on embryo survival and recruitment due to oil deposition in spawning areas.
    - There was no demonstrated avoidance of oiled areas by fish (anecdotal evidence only);
    - Fidelity to feeding and overwintering areas may have exposed the fish to oil in sediments and food organisms, or reduction in food abundance may have reduced fitness going into the winter season; and
    - Metabolizing the PAHs may have created an energy cost that affected fitness.
- Population-level effects.
    - Similar abundance and species composition of fish in snorkel surveys conducted in 1993 and 2005, 2006 and 2007 . H owever, the natural variability of abundance and the

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influence of migration from un -surveyed areas is unknown , and there were no meas ures of reproduction or recruitment.
    - No definitive statements can be made regarding the presence or absence of population - level effects.

8.6.6.4 *Spill response*

- Active recovery of liquid oil and oil -plus-soil from the river and spill site removed about 84% of the total spilled oil; 91% of the oil was either removed or accounted for.
- Physical removal from the river channel, shorelines and woody debris.
    - Direct damage to habitat from heavy machinery within the river channel;
    - Changes in channel morphology caused by ban k armouring cut off fish access to an important back-channel habitat area; and
    - Removal of logjams eliminated important fish habitat and altered flow dynamics, in some cases to such an extent that significant areas of side channel habitat were lost , and subsequent seasonal increases in flow restructured channels and degraded fish habitat

8.6.7 *Lessons learned from the Pine River spill*

- Fish kills in moderate -sized rivers can extend far downstream if the spill is not contained immediately.
- The extent of impacts on fish can be obscured if there is a slow response to the spill (e.g., remote spill site) and dead fish are consumed by scavengers before being identified.
- Benthic invertebrate communities can recover quite quickly after spills that occur in summer months, provided the initial scale of the impact is sufficiently small.
- The effects of cleanup activities were substantial, decreasing or eliminating important fish habitats.
- Data were insufficient to support any definitive statements regarding l ong-term effects on fish abundance and species composition.
- Opportunities for more detailed and sophisticated investigation of the effects of the spill on were not pursued, perhaps due to logistic and financial constraints.

8.6.8 *Research needs related to the Pine River spill*

<u>Recommendation</u>: Standard guidance is required for measurement of oil contamination of sediments, pore water and biota within flowing waters of Canada. This guidance should consider the range of lotic receiving environments and logistic constraints common t o remote locations. The guidance should include the *de minimis* level of investigation required for decision -making regarding cleanup requirements and techniques, as well as for determination of acceptable residual levels of oil.

<u>Recommendation:</u> The Pine Ri ver spill offered the opportunity to track longer -term response to cleanup activities, as well as to residual oil. Such opportunities should not be lost if such spills occur again, because the longer -term trade -offs involved in selection of cl eanup methods remain uncertain.

<u>Recommendation:</u> Standard guidance for monitoring the fate of spilled oil in all major environmental compartments of lotic systems is required.

<u>Recommendation:</u> Research is needed on the relative role of hyporheic flows in contributing to exposure of fish to hydrocarbons after a spill. A range of experimental conditions using relevant concentrations and both weathered and unweathered oil should be tested on salmonid species (and

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demersal spawners in general) under conditions that represent actual field conditions in terms of variables such as flow, temperature, substrate and dissolved oxygen.

<u>Recommendation:</u> The Pine River spill case study illustrates the importance of baseline information with which to compare post -spill monitoring data. The Panel strongly recommends the establishment of a national baseline database for freshwater systems adjacent to pipeline corridors (e.g., data collected for EIAs, environmental effects monitoring (EEM) p rograms, compliance monitoring [e.g. for municipal o r industrial discharges], research programs, etc). The Panel is aware that some databases are already established (e.g. for EEM data); however, there is no readily accessible, national portal to metadata on freshwater systems.

<u>Recommendation:</u> The Pine River spill could have yielded highly valuable data regarding effects of spills in riverine systems in the short, medium and long -term. The Panel recommends that there should be national guidance regarding 'spills of opportunity', by implementing communication and coordination protocols among agencies and industry to optimize the collection of scientific knowledge and lessons learned from incidents.

8.7 Wabamun Lake, Alberta Train Derailment

On August 3 rd, 2005, 43 Canadian National Railway Company (CN) rail cars derailed immediately adjacent to Wabamun Lake (**Figure 8.6**), approximately 65 km west of Edmonton, AB. The lake (area = 82 km^2; mean depth = 6.3 m; maximum depth = 11 m) is moderately to highly enriched with nutrients and is generally well mixed and thus generally well oxygenated throughout the water column during the open-water period (Prepas and Mitchell 1990; Hollebone 2008).

Of the 43 derailed train cars, 11 containing heavy fuel oil (HFO 7102, a type of Bunker C oil; Table 2.2) ruptured, spilling 712 m^3. A single car carrying Imperial Pole Treating Oil [6] (PTO) also ruptured spilling about 88 m^3 on to the ground at the derailment site. A total of about 149 m^3 of heavy fuel oil entered the lake (Bi rtwell 2008). The oils ran onto the lawns of cottages about 100 m from the lakeshore. HFO entered the lake less than 1.5 hours after the derailment along a broad front of about 0.5 km (**Figure 8.8**). The flow was aided by the fact that the HFO 7102 had been loaded a few hours before and was still warm and relatively less viscous than it would be at ambient temperature (Hollebone 2008).

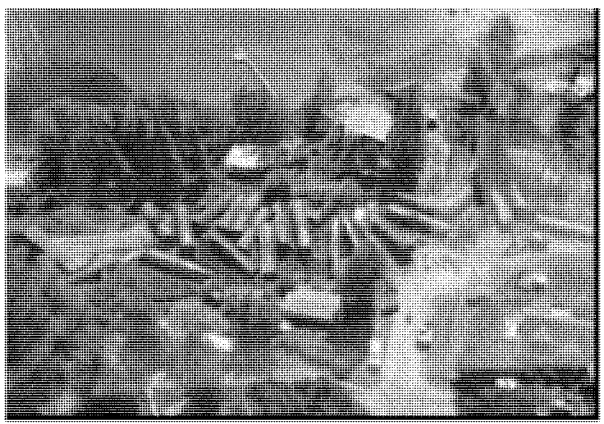

Figure 8.8 Wabamun Lake Train Derailment . Source: Dangerous Goods Newsletter, Spring 2006 . *Image from Transport Dangerous Goods Directorate, Transport Canada (https://www.tc.gc.ca/eng/tdg/newsletter-spring2006-323.htm)*

[6] *A hydrocarbon-rich liquid used for preserving wooden poles, containing naphthalene and other PAHs (http://www.enr.gov.nt.ca/sites/default/files/pahs.pdf)*

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8.7.1 Early behaviour and effects of the spill

Initially, all of the oil appeared to be floating on the surface, rapidly spreading in warm, calm weather conditions, and escaping boomed areas once the wind increased. Over 1.1 million m^2 of lake surface was visibly oiled, of which 63% was heavily coated (Birtwell 2008). A few days later, strong westerly winds and waves concentrated the oil along the north, east and south shorelines (Anderson 2005). Langmuir circulation patterns during the windy conditions contributed to the rapid horizontal movement of the oil, as well as vertical movement of oil droplets into the water column. Much of the shoreline was boomed off and oil was trapped with varying degrees of efficiency in the littoral zone (Anderson 2005; Hollebone et al. 2011). Measurement of hydrocarbon concentrations within the boomed area showed concentrations well in excess of guidelines for the protection of aquatic life. Concentrations of BTEX and PAHs in open-water areas of the lake outside of the boomed areas were well below water quali ty guidelines (Anderson 2005).

Tar balls quickly formed (from <1 to 10 cm in diameter) in near -shore areas. The rapid formation of tar balls was attributed to the loss of volatile components from the spilled oil which also picked up fine mineral and organic particles along its overland flow -path to alter its density (Parker -Hall and Owens 2006). Within hours, some of the tar balls were showing neutrally -buoyant behaviour, and were seen riding up and down in the water column. Some would rise to the surface, others would be seen sinking to the bottom. In addition to tar balls, tar logs were observed, up to 30 cm in di ameter and 5 m in length, consisting of a mixture of organic debris and oil (Hollebone 2008). Some near-shore areas had extensive tar mats.

Dead fish were observed in oiled areas immediately following the spill and for about two months afterward. A total of about 100 dead fish were observed by cleanup crews. Live fish captured two weeks after the spill showed biochemical evidence of exposure and had elevated levels of hydrocarbons in tissues (Hodson et al. 2007; Birtwell 2008).

People in the Whitewood Sands cottage community (where the derailment took place) were evacuated within 15 -20 minutes of the derailment and allowed back to their properties later that evening (Lake Wabamun Residents Committee 2007). The health authority advised residents on the day of the spill to avoid using the lake until further notice. In August, 2008, three years later, advisories were still being issued against eating certain wildfowl and the fishery was catch-and-release only.

8.7.2 Shorelines

During the Shoreline Cleanup and Assessme nt Technique (SCAT) survey in 2005, the majority of tar balls or tar mats were observed near the shores in water depths from 0.1 to 1.5 m, with their frequency decreasing with increasing depth (Hollebone 2008). After shoreline treatment a high proportion o f oil remained. The coverage of oil varied but was highest in treated reed beds (**Figure 8.9)**. In the reed bed areas on warm days, tar balls would rise to the surface, shed oil from several points around their circumference and create a sheen (Hollebone 200 8). Beach re-oiling continued until fre eze-up during the first winter.

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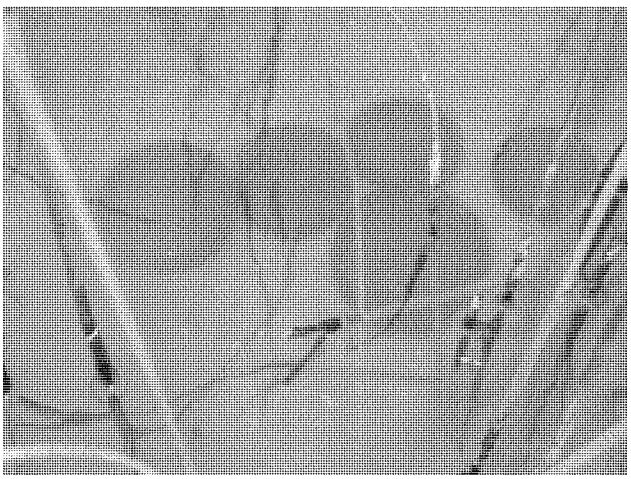

Figure 8.9. Near-shore reed bed with tar balls in Wabamun Lake. *Image from Hollebone (2008).*

By the spring of 2007, a SCAT survey showed almost all oiling to be in the Very Light or Trace categories found in marsh, peat-soil or vegetated bank shoreline types.

8.7.3 Weathering

Four months after the spill, about 79% of the HFO 7102 and 47% of the PTO had been recovered from the lake spill site, and flow path to the lake and water concentrations had decreased to below detection limits (Birtwell 2008). According to Birtwell (2008), acutely toxic PAH concentrations were likely limited to the reed beds, where high concentrations of PAHs in the water would be promoted because the water circulation would be poor and the surface area of the oil large.

While on the surface of the lake, the oil weathered considerably with a loss of most of the lighter components. Oil coating the reeds was heavily weathered 40 days after the derailment. The weathering of tar balls varied, with some tar ball samples collected about 80 days after release showing only moderate weathering. Some free -floating oil was able to survive until near freeze -up in a largely fresh, lightly -weathered state (Hollebone 2008).

A year and a half after the derailment in February 2007, oil was found in the lake in primarily two forms:

- Large (>5 cm) flat conglomerations on the lake bottom, often tangled into vegetation and highly weathered, with high sediment loading, water content and viscosity; and
- Small (<5 cm), less chemically-weathered spherical balls of soft, fluid oil containing a significant fraction of the original aromatic content of the fuel oil, surrounded by a tough, more weathered layer of oil, which were easily stirred up from the bottom and readily moved with currents and wind due to lower density and viscosity (Hollebone 2008).

Virtually all of the alkanes and 'Priority Pollutant' PAHs had attenuated by t he spring of 2007 (Parker -Hall and Owens 2007). Tar balls, particulates and the oil coating the vegetation contained primarily asphaltenes (which are unregulated chemicals) and low concentrations of non -'Priority Pollutant' PAHs (Parker-Hall and Owens 2007). Foght (2006) stated that while competent hydrocarbon -degrading microbes existed in lake sediment, the mass of oil they were likely to be able to degrade was small due to the recalcitrant nature of the oil in tar balls and coating the vegetation. Foght a lso pointed out that the tar balls would limit microbial access to the biodegradable components of the spilled oil and concluded that the prognosis for extensive natural biodegradation over 5-10 years was poor.

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Given the amount and chemical nature of residual oil in the spring of 2007, Parker-Hall and Owens (2007) concluded that further treatment or recovery of sunken oil beyond that planned for May and June of that year was not practical or feasible and would pose a risk of further damage to lake bottom and near-shore communities. Short (2008) estimated that cleanup activities recovered more than 95% of the oil components that were not lost through natural weathering.

8.7.4 Observed effects on aquatic biota

The chemistry of the heavy fuel oil spilled in Wabamun Lake differed in specific ways from typical Bunker C fuel oil and may have contributed to the observed toxicity, especially to fish embryos. The HFO 7102 product was primarily composed of saturated hydrocarbons and also contained a high content of aromatic hydrocarbons (48%) and high concentrations of PAHs (60,400 µg/g oil) (Hollebone 2008). In comparison, a typical Bunker C fuel has a total aromatic content of 29% and a total PAH content of 29,000 µg/g oil (Hollebone 2008). The PAHs of the HFO 7102 were predominantly alkylated naphthalenes (Table 2.1) with smaller amounts of alkylated 3-ring phenanthrene and members of the fluorene homologous series (Hollebone 2008). No BTEX compounds were detected except in the PTO.

8.7.4.1 Algae and macrophytes

Field observations of phytoplankton community characteristics and macrophyte survival and growth did not indicate any effects of oil exposure overall (Golder Associates 2007 cited by Birtwell 2008). The spill had little direct effect on the abundance and productivity of soft stem bulrush (*Schoenoplectus tabernaemontani)* (Thormann and Bayley 2008; Wernick et al. 2009) (see Chapter 4).

8.7.4.2 Zooplankton

Standard acute toxicity tests using the water flea species *Daphnia magna* and chronic tests using *Ceriodaphnia dubia* showed no acute lethality but reduced growth near the spill site. The reduced growth was considered to be a result of factors other than oil by Golder Associates (cited by Birtwell 2008b). However, Birtwell (2008) argued that the significant results of tests between reference and oiled locations in the lakes had validity even if background ambient conditions differed from those in the laboratory setting. The characteristics of the zooplankton community did not change when compared with pre-spill data (Golder Associates 2007 cited by Birtwell 2008).

8.7.4.3 Benthic invertebrates

No data existed on immediate effects of the spill on benthic invertebrates but physical smothering effects were considered likely (Birtwell 2008). Laboratory toxicity tests showed significant toxicity to two test species *Chironomus tentans* (a midge) and *Hyalella azteca* (an amphipod crustacean) from sediments collected near the spill site. However, the proportion of sand in the sediment was an important confounder, with higher mortality in high-sand substrates. Both test species showed significant reductions in growth in both high- and low-sand sediments. PAH concentration was correlated with toxicity to the nematode *Lumbriculus variegatus*, with significant reductions in growth in both high- and low-sand sediments.

8.7.4.4 Fish

Risk to fish was due to direct toxicity immediately after the spill, and longer-term direct and indirect risks associated with the near-shore areas, which were important for at least one life stage for the eight fish species inhabiting the lake. The near-shore areas were primary nursery, rearing and food supply areas, and fish did not avoid oiled areas. The summer timing of the spill increased the risk because it coincided with the presence of numerous life stages of each of the eight fish species present in the highly exposed areas (Birtwell 2008). Risk was also related to destruction of habitat during cleanup and survey activities.

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Hodson (2008) noted that the exposure of adult fish to the spilled oil corresponded to the discovery of significant numbers of dead fish in the oiled part of the lake. However, Hodson (2008) commented that heavy oils with low concentrations of LMW compounds would be less likely to cause acute lethality. The area closest to the oil spill was also affected by non-spill stressors, including: a thermal plume from a power plant; physical removal of macrophytes (to address concerns of cottagers regarding access to the lake for swimming and boating); and high fishing pressure (Schindler et al. 2004). Thus, fish inhabiting the area nearest to the spill were already exposed to other stressors and may have been more susceptible to the additional stress of the spill.

Within two weeks of the spill, a survey of adult fish demonstrated that exposure to PAHs from the spilled oil was widespread; this exposure was relevant with respect to sublethal effects (Hodson 2008). A second survey, about 10 weeks after the spill, showed continuing exposure of adult fish, although at a lower level than in August (Hodson et al. 2007; Hodson 2008;). One year after the spill, detectable PAH concentrations were found in fish flesh. However, PAHs in fish decreased over time due to metabolic processes in the fish plus weathering of the hydrocarbons in water and sediment (Birtwell 2008).

Embryo-toxic levels of PAHs were found in August 2005 through to spring of 2007. Toxicity studies conducted *in situ* by Golder Associates (2007 cited by Birtwell 2008) showed embryo deformities in caged eggs of lake whitefish (*Coregonus clupeaformis*) at spawning shoals in the winter and spring following the spill and on northern pike (*Esox lucius*) eggs caged in the spring of 2007. Effects were most pronounced in whitefish embryos with significant moderate to severe deformities correlated with PAHs in lake water months after the spill. Pike suffered >50% increased frequency of moderate to severe deformities relative to reference areas but few deformities overall (Golder Associates 2007 cited by Birtwell 2008; deBruyn et al. 2007; Hodson 2008). Uptake of PAHs into semi-permeable membrane devices deployed at the egg incubation site confirmed a correlation between PAHs and egg deformities.

Although some embryos survived exposure, the marginal state of a once-thriving whitefish population in Wabamun Lake raises concerns for cumulative impacts of oil and other anthropogenic stressors (Schindler et al. 2004; Donahue et al. 2006) on recruitment and production of whitefish. The whitefish fishery and all other sport fish in Wabamun Lake are catch and release only (Government of Alberta 2015).

8.7.4.5 Effects of the cleanup

Effects on fish habitat occurred due to the requirement to remove oil from Wabamun Lake (Evans 2008). These actions resulted in loss of structural habitat and alteration and disruption of lake substrates. Additional impacts occurred due to increased suspended sediment and turbidity during habitat removal and cleaning, increased erosion along shorelines that became unprotected from wave action, the remobilization of oil, and physical disturbance by trampling and similar activities.

Although the oil spill in Lake Wabamun did not have direct toxic effects on the reed beds, the 'treatment' impacts in the deeper reed beds reduced plant and rhizome density, and this may have had long-term consequences, remaining open or sparsely vegetated for several years and thereby providing lower quality habitat for fish.

Fish were killed when aquatic vegetation was cut. The presence of fish within harvested vegetation was probably less an issue of entrapment and an inability to escape than it was an example of their vulnerability due to habitat fidelity (the need for cover, etc.).

Benthic habitat and the associated communities were impacted due to the removal of oil by the use of numerous techniques, as well as the cutting of macrophytes and removal of whole plants. The techniques included low-pressure flushing, vacuuming and dragging over substrates. Trampling in shallow waters

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for survey work or for cleanup caused further damage. The impacts of the cleanup on the quality of habitat persisted for several years (Hollebone 2008) (see Chapter 4).

A survey of riparian habitat along the shoreline of Wabamun La ke in 2015 showed that approximately 57% of the lake's Riparian Management Area was in healthy condition, 9% was moderately impaired and 34% was highly impaired (North Saskatchewan Watershed Alliance 2015). Residential development was cited as the major cause of riparian disturbance.

8.7.5 Analysis of the response to the spill

In a review of the response to the spill, McCleneghan (2008) noted that containment of much of the oil within the first few hours of the spill would have been possible if the railway compan y had taken steps in the years preceding the event to prepare for an on-water spill from its operations.

CN estimated that it spent $28 million on cleanup and $7.5 million in compensation to property owners (Lilwal and Fitzpatrick 2015). The company was fined $1.4 million for its part in the disaster (Lilwal and Fitzpatrick 2015). The Wabamun spill spurred the establishment of a new agency in Alberta to coordinate disaster response. The Alberta Emergency Management Agency was created in 2006 (CBC 2006). According to the Agency website, it leads the coordination, collaboration and cooperation of all organizations involved in the prevention, preparedness and response to disasters and emergencies (http://www.aema.alberta.ca).

8.7.6 Public consultation and involvement

The spill generated outrage and protests, including a blockade over the CN tracks, with people accusing CN of being more concerned with restoring rail service than stopping the spread of the oil. The blockade ended after the company promised to meet with the public (Lilwal and Fitzpatrick 2015).

Ten years after the spill, some members of the public stated that the lake still suffered some lingering damage, with reports of tar balls and oil buried in the sand. However, a representative of the cottage community stated that CN had fairly compensated cottage owners and applauded the establishment of the Alberta Emergency Management Agency and stronger regulations for railways (Lilwal and Fitzpatrick 2015).

Public c onsultation was conducted by Alberta Environment, including workshops to discuss the establishment and operation of a Lake Wabamun Citizen's Panel that became the Wabamun Watershed Management Council, which provides input to watershed planning.

8.7.7 Aboriginal community concerns

Interaction between the Paul Band First Nation and provincial and/or federal government agencies was hampered due to the lack of notification, which resulted in challenges with respect to communication and collaboration. The First Nation filed a lawsuit over damage to the band's land and water. A settlement was reached with CN in 2008, resulting in a payment of $10 million (CBC 2008).

8.7.8 Important factors affecting the consequences of the Wabamun Lake spill

8.7.8.1 Oil properties and behaviour

- Wind dispersed the surface oil over a wi de area and onto shorelines . Langmuir circulation may have increased horizontal and vertical dispersion.
- The unique circumstances of the spill (hot oil flowing over land and picking up sediment before entering the lake) meant that tar balls were formed with accumulated sediment and sank.

- The heavy fuel oil contained about six times more total PAHs than most crude oils and also contained a higher proportion of alkyl PAHs than most crude oils (alkyl PAHs pose greater risk of chronic toxicity).
- The tough, weathered coating on the tar balls sequestered the less weathered oil within the balls, greatly limiting access to biodegradation and creating a longer-term source for release of less weathered (and more toxic) oil in near-shore environments as the tar balls broke apart.

8.7.8.2 Effect of the environment on fate and behaviour

- Lake morphometry and mixing characteristics.
  - The lake is shallow, wind-swept and well-mixed, which created conditions conducive to rapid surface dispersion; and
  - Shoreline characteristics that include small bays, creek mouths and marshy areas increased the areas with a high potential for accumulation of oil.
- Seasonality.
  - Winter conditions extended the time required for biodegradation and interrupted cleanup activities;
  - Less mixing energy under winter ice may have contributed to oil-related effects on fish already stressed by natural ecological factors; and
  - The behaviour of the residual oil in its various forms (e.g., tar balls) was influenced by temperature.
- Presence of existing anthropogenic stressors.
  - The area closest to the oil spill was also affected by a thermal plume from a power plant, physical removal of macrophytes (for aesthetic reasons) and high fishing pressure (Schindler et al. 2004). Thus, fish inhabiting the area nearest to the spill were already exposed to other stressors and may have been more susceptible to the additional stress of the spill.

8.7.8.3 Oil toxicity

- Relatively few fish deaths were observed, commensurate with the temperature of the oil when it entered the lake which was at summer temperatures, leading to rapid loss of low molecular weight components.
- There were possible effects on benthic invertebrate community structure but confounding factors, such as water depth and substrate particle size, may have been at least partially responsible for observed results.
- Sublethal toxicity:
  - Embryo deformities were observed in whitefish and pike associated with exposure to PAHs (the heavy fuel oil had a higher relative concentration of alkylated PAHs than typical Bunker C);
  - There was no avoidance of near-shore oiled areas by fish. Fidelity to nursery, rearing and feeding areas may have exposed the fish to oil in sediments and food organisms or reduction in food abundance may have reduced fitness going into the winter season; and
  - Metabolizing the PAHs may have created an energy cost that affected fitness and sufficient concentrations of PAH metabolites and reactive oxygen species to have caused oxidative stress (see Chapter 4).

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8.7.8.4 Spill response

- Physical removal of oil caused direct damage to habitat from trampling, low-pressure flushing, vacuuming and dragging over substrates and direct fish mortality due to being removed together with cut vegetation.
- Cleanup activities recovered more than 95% of the oil components that were not lost through natural weathering. The effects of physical removal were balanced against the degree of cleanup required to re-establish human uses of the lake.
- Delays in response were related to lack of preparedness and response capabilities.

8.7.9 Lessons learned from the Wabamun Lake spill

- The consequences of spills of oil adjacent to lakes can be greatly reduced if spill response is rapid and effective in preventing the oil from reaching the lake in the first place.
- Formal notification and involvement of Indigenous communities directly affected by oil spills is necessary for provision of ground support and to avoid adversarial relationships during the response.
- The exposure of spilled oil to soils and organic matter along the spill path increase the potential for production of tar balls.
- Deposition of tar balls and tar mats into heavily vegetated shorelines greatly increases the difficulty of cleanup and prolongs exposure of aquatic biota to physical smothering effects of oil, as well as to toxic effects of the oil.
- Cleanup of shoreline areas resulted in reduced density of vegetation and a decrease in habitat quality for several years.
- An advance plan should be developed for decisions regarding when the effects of cleanup activities outweigh the benefits.
- Wabamun Lake was one of the most well-studied lakes in Alberta, allowing a more confident assessment of the effects of the spill relative to baseline conditions (including effects of existing anthropogenic stressors).
- Crude oil spills into freshwater lakes can cause sublethal effects on fish which are observable in the field.
- There was a significant impediment to an assessment of impacts following the spill due to the priority given to enforcement activities and the control on collection and analysis of samples.

8.7.10 Research needs related to the Wabamun Lake spill

Recommendation: The Panel recommends a follow-up study of residual oils in reed beds and along sandy shorelines of Wabamun Lake. Anecdotal reports in 2015 of tar balls and buried oil in sandy beaches indicate the need for a study. This study would generate useful information, with respect to the chemistry and persistence of the residual oil, the impact of cleanup operations and the longer-term effects on ecosystem services, such as recreational use.

Recommendation: This spill represented a lost opportunity to learn about oil fate and effects in a small ecosystem where there was a high potential to discriminate oil effects from other natural and anthropogenic stressors. The Panel reiterates its earlier recommendation regarding identification of 'spills of opportunity'.

8.8 Spill from a Ruptured Pipeline into Talmadge Creek and the Kalamazoo River, Michigan

On July 25, 2010, a rupture in a pipeline carrying diluted bitumen (dilbit) resulted in the release of ~3,200 m^3 into Talmadge Creek and from there into the Kalamazoo River. It took over 17 hours for the rupture to be confirmed and the flow of dilbit to be stopped. The spill consisted of 23% Western Canadian Select

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and 77% Cold Lake Blend dilbit (Table 2.2). Western Canadian Select is a heavy blended unconventional sour crude (*CrudeMonitor.ca*) composed of bitumen blended with sweet synthetic and condensate diluents and 25 streams of conventional and unconventional Alberta hea vy crude oils blended at the Husky Terminal in Hardisty, AB. Cold Lake Blend is an asphaltic heavy crude bl end of bitumen and condensate.

The spill occurred during a period of high rainfall . W ater-soaked soils allowed the dilbit to easily run overland to Talmadge Creek and from there downstream to the Kalamazoo River. The flood flows caused significant contamination of riparian lands. The flow conditions had an exceedance probability of 4% with a mean velocity of about 1.1 m/s and a mean depth of 1.2 m (Fi tzpatrick et al. 2015). About 3.2 km of Talmadge Creek, 60 km of the Kalamazoo River and three impoundments involving medium to high quality wetlands were affected (Noble 2012). The spilled dilbit ultimately reached a reservoir called Morrow Lake about 65 km downstream. At that point, the high flows dropped, st randing oil on the floodplain.

Local residents self -evacuated from 60 residences and about 320 people reported symptoms consistent with crude oil exposure, including headache, nausea and respiratory s ymptoms (Stanbury et al. 2010; EPA 2012; Fitzpatrick et al. 2015). Benzene (Table 2.1) was the primary public health concern for residents and workers during the first 30 days.

About 2,900 m^3 of the spilled oil was recovered during the first year and about 10% submerged (about 300 m^3) (Fitzpatrick et al. 2015). The diluent of natural gas condensate volatilized. The weathered bitumen was positively buoyant at room temperature but submerged when mixed with river sediment under natural turbulent river conditions where it caused persistent globule and shee n releases (Fitzpatrick 2014).

8.8.1 Oil-particle interactions and submergence

The spill illustrated the importance of understanding the factors contributing to the formation and submergence of oil particle aggregate s (OPAs; Figure 2.7). Flood conditions at the time of the spill increased turbulence and the presence of suspended particulate matter that, in turn, associated with oil particles to form OPAs (Fitzpatrick et al. 2015). Additional mixing from flows over two dams may also have played a role, although OPA and submerged oil accumulated in the first 5 km of river length, between the spill source and the first dam.

Some additional features of the Kalamazoo River may have been important factors in OPA formation, transport and deposition (Fitzpatrick et al. 2015). These features included:

- Abundant wetlands in the floodplain and flooding of riparian lands at the time of the spill . Thus, suspended and bottom sediments had relatively high organic matter content;
- The river is wide and has an average gradient of 0.06% in the spill-affected reach;
- Channel margins, backwaters, side channels, oxbows and impoundments all provided depositional areas; and
- Post-spill high-flow events caused re-suspension and resettling of OPA in downstream areas.

The USGS (Fitzpatrick et al. 2015) suggested that for impoundments with accumulations of thick fine - grained sediment, the process of gas bubble formation (e.g., from microbial methane production in anaerobic sediments) and release from s ediments (ebullition) is likely to be an important mechanism for re-suspending OPAs in the water column and releasing oil as sheen on the water surface. Spontaneous releases of oil globules and floating OPAs have been observed regularly in the impounded se ctions of the Kalamazoo River during 2011–14, resulting in oil sheens at the water surface.

Fitzpatrick et al (2015) noted that OPA formation is a natural process that enhances the physical dispersion of oil and might result in enhanced biodegradation (Figure 2.3). The authors noted that the formation of OPAs would reduce the bioavailability and toxicity of the residual oil to aquatic organisms. They suggested that active enhancement of OPA production could be an alternative to the use of chemical dispersants. However, the prescribed sinking of spilled oil or the use of sinking agents is currently prohibited by the U.S. EPA because of the potential risks of acute and chronic toxic effects on benthic organisms and possibly decreased biodegradation once th e oil is deposited and buried in anaerobic sediments (Fitzpatrick et al. 2015). These risks were illustrated in the case of the Wabamun Lake spill, where the tar balls or tar mats formed from interaction with soil particles were resistant to biodegradation and prone to releasing relatively unweathered oil from the interior of the tar balls. Tar balls and tar mats also continue to come ashore in the Gulf of Mexico subsequent to the DWH spill.

Fitzpatrick et al. (2015) concluded that a major factor in ecologi cal risk of OPAs is whether the y are physically diluted in suspension (less risk) or concentrated by deposition (more risk). The added context of water depth and the geographic extent are also important contributors to risk.

8.8.2 Environmental effects

Limited acute toxicity testing was performed on oiled sediments from the Kalamazoo River. As part of a NEBA, effects on aquatic organisms from weathered oil were assessed in laboratory acute toxicity studies of seven sediment samples collected from oil -affected backwater habitats along the Kalamazoo River in February 2012, about 19 months post -spill (Bejarano et al.2012 cited in Fitzpatrick et al. 2015). Ten -day whole sediment toxicity tests used *Chironomus dilutus* and *Hyalella azteca* and included survival, growth and biomass as the toxicity endpoints. Results from the toxicity tests indicated that *C. dilutus* were more sensitive to oiled sediment (and presumably OPAs) than *H. azteca* but that all samples exceeded the minimum survival (70%) and growth (0.48 mg ash -free dry weight at test termination) criteria for acceptable controls for the *C. dilutus* tests.

Fitzpatrick et al. (2015) concluded that on the basis of the weight -of-evidence approach and additional risk metrics, it is possible that residual oil at two heavil y and one lightly oiled area may pose some risks to benthic receptors . H owever, chronic toxicity of residual oil to sensitive life stages of benthic invertebrate and fish species remained unknown at the time of writing (January 2015).

In the fall of 2010, all locations on Talmadge Creek and the Kalamazoo River showed signs of impact, but it was unclear whether the impacts were from the spilled dilbit or from physical disturbance caused by cleanup activities (see discussion of cleanup below). Benthic inverte brate abundance and diversity were reduced in Talmadge Creek, and habitat disturbance was severe (particularly overbank areas) (Noble 2012). Abundance and diversity in the Kalamazoo River were reduced due to severe habitat disturbance, sedimentation problems and bank erosion. By the summer of 2011 (13 months after the spill) benthic invertebrate diversity increased in areas of Talmadge Creek altered by the cleanup. Sheen and oil were present. The creek channel was more open, with more sunlight penetration due to extensive removal of oiled vegetation. Abundance and diversity of benthic invertebrates had also improved in the Kalamazoo River, but ongoing work continued to cause impacts. Increased sediment movement and deposition were problematic, and sheen and oil were noted during surveys. In August 2012, a second round of response work disturbed 3 km of the creek. However, according to Noble (2012) the restored creek provided much better habitat compared to 2011.

No fish kills were observed although the Talmad ge Creek fish community was reduced and habitat greatly diminished in 2010 with some recovery in 2011 (Milsap et al. 2012). Some declines in fish community diversity and abundance were observed at some sites in the Kalamazoo River.

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Crushed and freshly dead mussels were found in the spill area but not in a reference area. Over 3,000 turtles, 170 birds and 38 mammals were brought to rehabilitation centres with survival rates of 97%, 84% and 68%, respectively (Milsap et al. 2012). Turtles were affected the most because they burrow down into sediments. The affected sections of the Kalamazoo River were closed to publi c access for nearly two years.

8.8.3 Effects of cleanup activities

Talmadge Creek was reconstructed due to removal of bitumen deposits (**Figure 8.10**). Significant areas of riparian vegetation were stripped from the banks of Talmadge Creek and riparian lands converted to dirt roads.

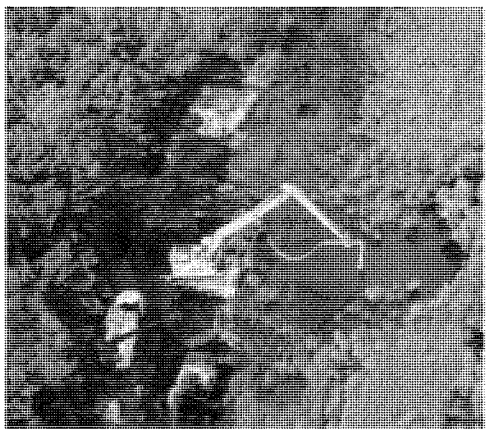

Figure 8.8 Response operations near the source of the spill on Talmadge Creek. *Image from US EPA.*

The persistent appearance of an oil sheen on the water was a cause for concern among the public, spurring efforts to locate, dislodge and disperse submerged oil (the source of the sheens) (**Figure 8.11**; Chapter 6). This activity employed long poles inserted into sediment to detect the submerged oil, followed by agitation to release buoyant oil, which continued until there was no sheen (S. Hamilton pers. comm.). Aggressive sediment agitation techniques (raking, flushing, aeration and skimming the river bottom physically or with water jets) was conducted in 2011 to liberate submerged oil as recoverable sheen, potentially contributing to further OPA formation and transport of OPAs to downstream reaches (Fitzpatrick et al. 2015). This was illustrated by results b efore and after agitation a t the Ceresco Dam (Noble 2012).

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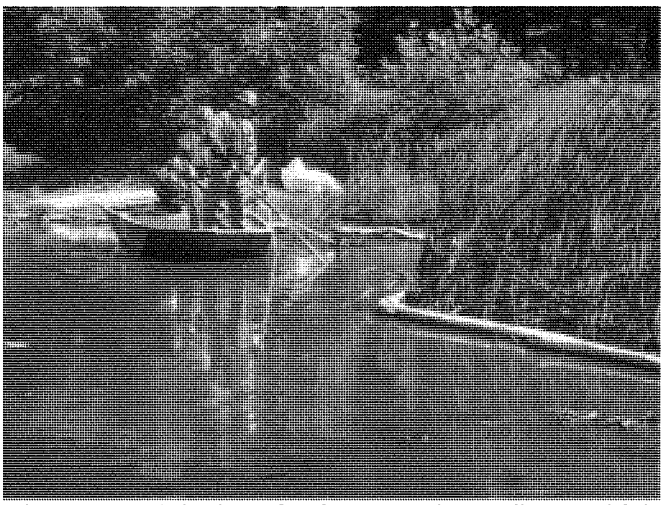

Figure 8. 11 Agitation of Kalamazoo River sediment with jets of water to flush submerged oil to the surface . *Image from US EPA.*

The persistent residual submerged oil and oiled sediment in the Kalamazoo River resulted in a protracted cleanup that ultimately required dredging and removal of several islands (because they had been over-washed by oil) and has accounted for a major share of the cleanup costs, which to date have surpassed $1.2 billion (Fitzpatrick et al. 2015).

An advisory group established to evaluate the risks and benefits of cleanup concluded that natural biodegradation could not be relied upon in this case, primarily because heavy oil is chemically recalcitrant to biodegradation and furthermore became buried in anaerobic sediment. The group noted that the weathered submerged oil had low acute toxicity and suggested that the majority of the submerged oil should be left alone. However, there were specific reservoir areas that were identified for further dredging. The group noted that agitation should no longer be considered an option since less than 1% of the oil was released and potentially further degraded the habitat (S. Hamilton, pers. comm.).

8.8.4 Evaluation of the spill response

The National Transportation Safety Board (NTSB) (2012) noted "pervasive organizational failures" in the pipeline company that included:

- Deficient pipeline integrity management procedures, which allowed well-documented crack defects in corroded areas to propagate until the pipeline experienced catastrophic failure;
- Inadequate training of control centre personnel, which allowed the rupture to remain undetected for 17 hours and through two start-ups of the pipeline that increased pressures and the amount of oil lost; and
- Insufficient public and agency awareness and education, which allowed the release to continue for nearly 14 hours after the first notification of an odour to local emergency response agencies.

The NTSB (2012) stated that the following contributed to the severity of environmental consequences:

- The pipeline company's failure to identify and ensure the availability of well-trained emergency responders with sufficient response resources;
- The lack of regulatory guidance for pipeline facility response planning by the Pipeline and Hazardous Materials Safety Administration (PHMSA); and
- PHMSA's limited oversight of pipeline emergency preparedness that led to the approval of a deficient facility response plan.

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8.8.5 Important factors affecting the consequences of the Kalamazoo River spill

8.8.5.1 Oil properties and behaviour

- Rapid evaporation of the diluent occurred, creating a respiratory and combustion hazard.
- There was a consequent increase in density of the weathered dilbit.
- Biodegradation potential was limited due to the high proportions of HMW compounds, such as resins and asphaltenes.
- Abundant wetlands with high organic matter contributed to OPA formation, followed by submergence.

8.8.5.2 Effect of the environment on fate and behaviour

- River flow.
  - Flood conditions at the time of the spill increased turbulence and the presence of suspended particulate matter that, in turn, associated with oil particles to form OPAs, and caused oiling of riparian lands and vegetation; and
  - Post-spill high-flow events caused re-suspension and resettling of OPA in downstream areas.
- River channel characteristics.
  - Channel margins, backwaters, side channels, oxbows and impoundments provided depositional areas.
- Sediment characteristics and processes.
  - Gas bubble formation and release from fine-grained sediments in reservoirs (ebullition) re-suspended OPAs, resulting in a sheen on the water surface; and
  - Fine-grained sediment reduced oxygen replenishment and led to anaerobic conditions.

8.8.5.3 Oil toxicity

- There were few apparent acute toxicity effects—no fish kills were noted immediately after the spill and there was high survival among oiled turtles and birds.
- There were possible effects on benthic invertebrate community structure but effects of cleanup activities may have played a greater role than oil toxicity; e.g. crushed and dead mussels.
- OPAs appeared to be low in toxicity (at least to two benthic invertebrates in laboratory toxicity tests).
- There was no publicly available information on sublethal toxicity to fish.

8.8.5.4 Spill response

- Delays in response were related to lack of preparedness and response capabilities, as well as to deficiencies in regulatory oversight.
- Physical removal of oil caused:
  - Direct and major damage to habitat along Talmadge Creek. However, reclamation appeared to have successfully produced viable habitat;
  - Apparent direct mortality to mussels (via crushing) from cleanup activities; and
  - Agitation to dislodge submerged oil and create recoverable sheen appeared to cause greater impacts as it resulted in negligible oil recovery and most likely dispersed the oil into deeper anaerobic layers of the sediment where it would become more persistent.

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8.8.6 Lessons learned from the Kalamazoo Spill

- Spilled dilbit can form OPAs under conditions of turbulent flow with elevated suspended sediments and abundant organic matter along the spill flow-path.
- Acute effects from spills of dilbit can be minimal given rapid evaporation of diluents.
- Chronic effects of residual bitumen can be difficult to distinguish from the effects of cleanup activities; therefore, a careful evaluation of net environmental benefit is required.
- It may be best to rely on (slow) natural cleanup process es for submerged oil since the negative effects of cleanup may be significant . H owever, the NEBA may also have to include net sociological benefit because of concerns of the public.
- The costs of cleanup can be huge relative to demonstrated effects . Cleanup appeared to be driven by concerns over submerged oil even though effects from the submerged oil were not particularly apparent (apart from the aesthetic concerns raised by sheens).

8.8.7 Research needs related to the Kalamazoo spill

Recommendation: Investigation of OPA behaviour and effects on habitat quality and on benthic invertebrates and fish would contribute valuable information to the overall evaluation of risks of spills of dilbit. Various approaches could be used to represent Canadian receivi ng environments, including a combination of laboratory and mesocosm experiments. Experimental conditions which include ice are required for the Canadian context.

8.9 Other Recent Spills

8.9.1 Westridge 2007 Spill, Burnaby, British Columbia

On July 24, 2007, a backhoe operated by a third -party contractor accidentally ruptured the Trans Mountain pipeline carrying crude oil to the Westridge Martine Terminal. Crude oil from the punctured pipeline sprayed about 12 to 15 m into the air for about 25 minutes. Fifty homes and properties, as well as a section of the Barnett Highway , were affected (TSB 2007). Approximately 234 m^3 of heavy synthetic crude oil blend was spilled. Approximately 40% of the oil entered the storm drain system and reached Burrard Inlet through shoreline storm outfalls, a submerged storm outfall and Kask Creek. Once in Burrard Inlet, the oil began to spread further into the inlet through wind and tide action (TSB 2007).

Western Canada Marine Response Corporation (WCMRC) responded to the spill within an hour and began booming marine areas within 45 minutes of oil being discharged through the shoreline storm water outfall (Stantec 2012).

Emergency phase cleanup techniques included:

- Booming to contain oil around the release points and also exclusion booming to pr otect sensitive shorelines; and
- Skimmers and absorbent pads to remove oil.

According to Kinder Morgan (2014), a pproximately 95% (210 m^3) of the released oil was recov ered. An estimated 5.5 m^3 was not recovered and was considered to be released to the marine env ironment (Kinder Morgan 2014).

SCAT was used to identify oiled shorelines, establish cleanup methods and set priorities. Approximately 1,200 m of shoreline were affected by the spill (TSB 2007). Recovery and rehabilitation of affected wildlife was performed through daily surveys for six weeks following the spill (Stantec 2012). By 2011, most of the recovery endpoints established for the spill had been met (water q uality in 2007; intertidal sediment quality, PAH levels in crab and intertidal community structure in 2011) (Stantec 2012). In 2012,

PAH levels in mussels had not yet met the endpoint . However, results were considered to be confounded due to other sources of PAH in the area (e.g., urban runoff and vessel traffic) (Stantec 2012).

Favourable environmental factors at the time of the spill included:

- Sunny weather (no rainfall runoff to increase movement of oil in storm drains , good evaporation conditions);
- Slack tide conditions which helped keep the oil near the shore while booms were placed;
- Timing which was of primary migration and overwintering period for birds and after the breeding bird season; and
- Timing was prior to the main salmon migration period (Stantec 2012).

8.9.1.1 Lessons learned

The Westridge Pipeline spill was a case where spill response was appropriate and successful.

The TSB (2007) determined the factors contributing to the third -party pipeline breach. These factors are listed below:

- The field location of the Westridge Pipeline was not accurately indicated on design drawings, and the location was not verified, as required under the NEB *Pipeline Crossing Regulations, Part I* . Therefore, the actual field location was not discovered before the start of construction;
- Inadequate communication within Kinder Morgan Canada, Inc., and between Kinder Morgan, the consultant and the contractor;
- The conditions of the crossing agreement and the NEB *Pipeline Crossing Regulations Parts I and II* respecting an onsite pre -construction meeting, locating the pipeline and supervision of construction activities were not adhered to, thus compromising the safe operation of the pipeline; and
- The initial decision by Kinder Morgan to shut down delivery pumps without isolating the gravity feed from the terminal instead of continuing with drain -down of the Westridge Dock Transfer Line increased the volume of crude oil released and was not in conformity with standard emergency shutdown procedures (TSB 2007).

A review of the incident by Kinder Morgan led to the implementation of a Pipeline Protection Department with responsibility for:

- Public awareness;
- Pipeline and associated facilities markings;
- Permits for safe work around pipeline and associated facilities;
- Aerial and ground patrols; and
- Response to emergency calls (Kinder Morgan 2014).

The WCMRC noted that there would be a clear benefit of having an on -water facility linked closely to personnel access and stated that it is reviewing different options within Burrard In let to achieve that end (Kinder Morgan 2014).

8.10 Release of Fuel Oil from the *Marathassa* Grain Carrier into English Bay, British Columbia

On April 8, 2015, about 2.7 m^3 of intermediate fuel oil (suspected to be IFO 380) was leaked from the *Marathassa* bulk grain carrier into English Bay, Vancouver (**Figure 8.12**) (CCG 2015a). This resulted in highly visible consequences due to the multi-use nature of the English Bay area (anchorage for Port Metro Vancouver tanker traffic, tourism, recreation). For example, the spill formed a visible sheen and fouled

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beaches around English Bay, the North Shore, Stanley Park and up into Burrard Inlet, resulting in closing of beaches by Vancouver Coastal Health. One week after the spill, the DFO closed recreational fisheries in a section of the affected area after advice from Vancouver Coastal Health. The Musqueam First Nation issued its own urgent notice one day after the spill, warning those who harvest crab and prawn in the area to stop fishing. However, subsequent chemical analyses of water samples showed hydrocarbon levels below laboratory detection limits, including in waters surrounding the ship, in line with this very small and localized spill.

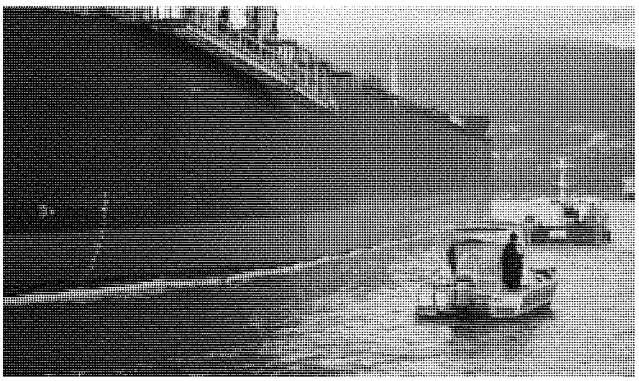

Figure 8. 12 Booms placed around the Marathassa. *Image from CCG (2015a) (http://www.ccg-gcc.gc.ca/independent-review-Marathassa-oil-spill-ER-operation)*

An operational update by the Canadian Coast Guard on April 23, 2015 (CCG 2015b) reported that shoreline cleanup and assessment teams had completed work, with no observed oil reported. Beaches were re-opened at all affected beaches, in conjunction with Vancouver Coastal Health. Sediment and mussel sampling was reported to be ongoing and the precautionary recreational fisheries closure remained in effect. An independent review of the spill response provided to the Canadian Coast Guard in July, 2015, noted that impact on the public was minimal from a health and safety perspective. However, Environment Canada estimated that approximately 20 birds were affected. As of the date of this writing, monitoring continues.

8.10.1 Lessons learned

An independent review of the spill response (CCG 2015a) noted a number of areas for improvement in spill response:

- A need for improved communication with partners to ensure accuracy of communications;
- The Canadian Coast Guard did not have sufficient nearby initial capacity to respond due to demobilizing from another pollution response. Therefore, the Canadian Coast Guard contracted the WCMRC to initiate the on-water response and provide support to the Incident Command Post;
- Information sharing on a common network was not optimal due to federal government electronic policies and protocols;
- Canadian Coast Guard had not yet reached full operational capacity for its Incident Command system. Therefore, it took several days for Unified Command to achieve an operational rhythm;
- Early alerting of municipalities, First Nations and stakeholders was delayed due to the low classification of the incident in the provincial alerting system;
- Vancouver Area Emergency Response Planning timelines did not align with the immediate need to engage partners in the development of an efficient and effective plan in Vancouver harbour;
- The lack of a physical presence of Environment Canada impacted the effectiveness and efficiency of the Environmental Unit; and

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- Public communications from Unified Command were challenging because energy was focused on supporting government officials in media briefings, rather than ensuring key facts were being shared with citizens and Unified Command partners.

The independent review presented 25 recommendations for consideration by the Canadian Coast Guard and partners. The recommendations covered communication, training, standards, adequacy of staffing, systems, protocols, tools, approval processes and need for involvement of other federal agencies, notably Environment Canada (CCG 2015a).

In addition to the findings of the independent review, a representative of the Vancouver Aquarium (one of the groups that assisted in the response) pointed out that:

- Gaps exist in research and preparedness because of the lack of cohesive long-term monitoring of coastal ecosystems, creating a lack of baseline data which, in turn, makes it difficult to assess spill impacts;
- There is no official clarity around who is to monitor the effects of a spill; and
- Coordination among agencies and organizations was lacking, resulting in duplication of effort (e.g. at least three agencies collected water samples for testing) and/or gaps in information (Kane 2015).

Recommendation: The Panel trusts that the recommendations made to the Canadian Coast Guard by the independent reviewer arising out of the *Marathassa* spill will be fully implemented. The Panel notes that deficiencies in communication and coordination are common themes among the Canadian case studies reviewed in this chapter.

Recommendation: The Panel reiterates its recommendation for standard guidance regarding monitoring after oil spill events, as well as a targeted program of baseline data collection focussed on areas identified as high-risk.

8.11 Release of Bitumen Emulsion from a Pipeline in Alberta

On July 15, 2015, Nexen Energy reported a pipeline failure at its Long Lake facility near Fort McMurray, AB, that released an estimated 5,000 m^3 of an oil emulsion consisting of bitumen, process water and sand. The release affected an area approximately 40 m by 400 m and was confined primarily to the pipeline right of way, which includes muskeg (AER 2015a). A water body near the release site had apparently not been impacted. The spill created concern because of its size, and the fact that it involved the rupture of a new, double-walled pipeline that had recently been inspected. There were also reports of at least one waterfowl death.

Subsequent to the release and investigation by the Alberta Energy Regulator (AER), it issued a Suspension Order to Nexen due to what was referred to as "noncompliant activities at Long Lake oil sands operations pertaining to pipeline maintenance and monitoring" (AER 2015b). The order directed the company to: immediately suspend 15 pipeline licenses, which required shutdown of 95 pipelines carrying natural gas, crude oil, salt water, fresh water and emulsion; and provide sufficient documentation to assure the AER that the pipelines can be operated safely. As of this writing (September 2015), the suspension has been partially lifted. The cause of the failure is under investigation.

8.11.1 Lessons learned

This spill illustrates that the technology incorporated into newer pipelines is not sufficient to eliminate failures, even of monitored double-walled pipelines. New pipelines must be managed, maintained and monitored, and their performance must be reported rigorously and transparently.

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This case also illustrates that the AER apparently was not aware of Nexen's lack of compliance regarding maintenance and monitoring until after the spill occurred. Regulatory oversight is required at all times, not just after a spill event.

Recommendation: Systematic and efficient regulatory oversight of industry inspection, testing and maintenance is required for all pipelines, old and new.

8.12 Overall Conclusions Arising from the Case Studies

Delayed response was a critical factor affecting the consequences of all of the oil spill case studies. In some cases, adverse weather was a factor, which delayed initial response. Remote locations made quick response difficult. Ineffective well-capping technology resulted in a flow of oil from the DWH blowout, which lasted three months (although flow was greatly reduced two months after the blowout). However, notwithstanding the importance of weather, remote locations and technological challenges, human error (at an individual and organizational level) was a dominant factor across all case studies. Absent or inadequate planning, insufficient integration, inadequate training, poor communication, insufficient capacity (personnel and equipment), poor or no information sharing and lapses in regulatory oversight were noted for most, if not all, spill case studies. Despite the improvement in engineering design and monitoring technology, spills continue to go unnoticed and unreported for unacceptable lengths of time (e.g., the recent Nexen pipeline spill).

The case studies illustrate that the **consequences of oil spills cannot be predicted simply on the basis of the type of crude oil.** As discussed in Chapters 2, 3 and 4, the interactions among oil chemistry and the receiving environment produce a wide range of potential exposures to aquatic organisms, with an associated wide range of toxicity to individuals and effects on populations. Thus, the effects of diluted bitumen spills will not always be more severe than spills of conventional crude oil. For example, the spill of light crude oil into the Pine River, BC, had more severe consequences (particularly with respect to acute mortality) than the spill of diluted bitumen into the Kalamazoo River.

Studies of the medium and long-term effects of spills into fresh water are rare to non-existent. The consequences of crude oil spills to freshwater systems in Canada can be substantial because of less dilution potential and because of the tendency for oil to quickly become stranded along shorelines where it can remain for long periods of time. It cannot be assumed that spills in fresh water cause fewer adverse effects than spills in marine systems.

Relevant baseline data are often limited or completely lacking, thus hampering rapid decision-making regarding the most appropriate spill response and cleanup measures and limiting the ability to design an appropriate monitoring program with sufficient statistical rigour to reliably indicate effects. Data should include key species at risk by region and type of watercourse (marine offshore, marine inshore, estuaries, rivers, lakes, wetlands). Studies should include analysis oil biodegradation potential to assist in the prediction of environmental persistence as well as the development of remedial technologies.

There is often inadequate knowledge of the natural variability of indicators used in post-spill monitoring. Therefore, achieving a rigorous statistical sampling design capable of establishing whether there are significant effects from an oil spill is challenging. The effects of confounding anthropogenic factors are also often not well understood (e.g., other point and non-point hydrocarbon sources, the presence of other physical and chemical stressors, invasive species, commercial or recreational fisheries and habitat degradation caused by human activities). The influences of large-scale phenomena, such as the Pacific Decadal Oscillation and climate change, on biological indicators must also be considered.

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SL Ross (2014) provided the following suggested requirements for baseline information in support of monitoring effects on fish:

- Species of potential interest in the area threatened by the spill (e.g., sentinel species, most abundant species, protected species, exp loited species, i.e., subsistence, commercial, recreational);
- Abundance and spatial distribution of key species within and beyond the areas threatened by oil;
- Location of nearby areas that could serve as uncontaminated control areas for comparative purposes (e.g., similar habitat, same species, age distribution, conditions, physical conditions); and
- Information concerning baseline levels of biomarkers in key species prior to the spill (e.g., tissue levels of PAHs, EROD levels, information on spatial or seas onal variation in biomarker parameters).

In addition to the above list, there is a need for baseline information on measurements that allow assessment of effects on productivity in aquatic systems, particularly fish productivity. For example, data on fish age and size distribution and recruitment of young would provide the basis for evaluating whether a spill had the potential to alter the population via partial elimination of a year class.

Priorities for the acquisition of baseline data can be based upon t he identification of locations that are under the greatest risk of contamination from oil spills. The results of existing national relative risk assessments (such as those reviewed in the following section) can be used for prioritization. A review of locat ions with a combination of characteristics that increase risk should be conducted. These characteristics could include increased likelihood of exposure to oil spills due to proximity to multiple modes of transport of crude oil (e.g. rivers with pipeline cr ossings, railways and/or roads running parallel to the river, and oil tanker traffic in estuarine areas). Other characteristics could include relative sensitivity of the receiving environment determined by the presence of critical habitat and/or sensitive species, and the presence of culturally important species. Professional judgement will also be needed for this prioritization. Furthermore, it may be more appropriate to focus on indicators , such as benthic invertebrates, for standard monitoring purposes.

Aquatic ecosystem resilience to shocks from oil spills is key to the maintenance of ecosystem services. The presence of long-term stresses may tax the capacity of ecosystems to be resilient to shorter-term shocks. In some cases, ecosystems will undergo fun damental shifts in structure and function following disturbances (e.g., changes in trophic structure caused by invasive species). If policies and decisions are to be based upon resilience and the preservation of ecosystem services, multi -disciplinary research effort on the resilience of key marine and freshwater ecosystems will be required.

Effects of oil spill cleanup on aquatic ecosystems can be significant. This is particularly true for cleanup of stranded oil in both marine and freshwater environments, whether it is in the inter -tidal zone, the shorelines of lakes and rivers, or on woody debris and man -made structures in river channels. A careful, but rapid, NEBA is required tailored to each unique spill situation. Unfortunately, there is insufficient kn owledge of the long -term consequences of residual stranded oil to allow for confident NEBA in many environments, including Arctic and sub-Arctic systems.

The potential for long -term effects on animal populations continues to generate much debate and controversy. The Panel recommends that there be support for long -term research into effects of different oil types on populations of aquatic biota, especially fish, marine mammals and waterfowl. This research could be part of the suite of studies associated with significant oil spills.

There is an urgent need for the development and production of innovative, cost -effective and readily available spill prevention and response measures for use in Canada, including the Arctic. As

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already stated in this report, if spills do occur, maximum advantage should be derived from the opportunity to study the fate, behaviour and effects of the spill in the short, medium and long -term. Studies of the relative effectiveness of response measures should also be part of a suite of investigations associated with all significant oil spills in Canada. Pre-approved funding should be put aside for 'spills of opportunity' that would incorporate a combination of research and monitoring. Responsibility for each component of spill studies must be clear and research and monitoring efforts must be well coordinated to eliminate redundancy and to ensure that all required study components are implemented.

The consensus of scientists in oil spill countermeasure research is that major advances in the development and validation of spill response technologies are being hampered by our inability to conduct controlled field experiments with oil. In addition, validation is hampered by crude or no methods for measuring the efficacy of response measures. While 'spill of opportunity' case studies will provide lessons learned, they do not provide an optimal platform for the delivery of science due to logistical constraints (e.g., lack of experimental replication due to site heterogeneity, etc.). To support predictive numerical models and operational guidelines for spill response, there is a need for a rigorous database for the fate, behaviour and effects of various types of oil spilled and the efficacy of current and emerging oil spill countermeasures over a range of environmental conditions. This will require support for the conduct of field trials designed with statistical rigour that incorporate controlled releases of oil.

National guidance for monitoring of oil spills is urgently required to ensure that information gathered is reliable, adequate, credible and consistent. The guidance should include provisions for adjustment in response to specific characteristics of the receiving environment. The guidance should also include requirements for standard baseline datasets (tailored to specific receiving environments). The guidance could be divided into two parts: 1) which information to collect without exception; and 2) which information gathering can be deferred until the full scope of the spill and its potential effects are better understood. The guidance should include information on data quality requirements, including determination of minimum sample sizes, standard sampling protocols and laboratory quality assurance/quality control.

SL Ross (2014) also provided a list of recommendations for specific areas of research and development that can improve the ability to mount monitoring efforts efficiently for spills in Canada. The list is as follows:

- For potential Canadian biomonitoring species (cod, Arctic cod, snow crab, salmon, flatfish, mussels, oysters) prepare an annotated bibliography of studies on effects and relationships among exposure, bioaccumulation of and effects of PAHs (e.g., physiological biomarkers). *The Panel adds specific mention of freshwater species, such as whitefish, pike or small-bodied fish species, used in programs such as the national Environmental Effects Monitoring program.*
- For representative species used for monitoring, determine the exposure-response-recovery relationships for biomarker activity and occurrence of histopathological changes. *The Panel suggests that the relevance of physiological biomarkers to exposure versus effects should be examined (see Chapter 4).*
- Identify and acquire or develop a model for estimating fate, behaviour and persistence of spills into rivers. *The Panel notes that several hydrodynamic models are already available that are in common use in Canada. However, very few are three-dimensional and there appear to be no widely accepted models to describe surface water-groundwater interactions across a wide array of river types.*

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8.13 Prediction of Risk of Crude Oil Spills in Canada

8.13.1 Risk assessment for marine spills in Canadian waters south of 60th parallel

A risk assessment of marine spills along four sectors of the Canadian coastline - the Atlantic and Pacific Coasts, the Estuary/Gulf of St. Lawrence and the Great Lakes/St. Lawrence Seaway System - was conducted for Transport Canada and reported in 2014 (WSP and SL Ross 2014a), hereafter referred to as the TCRA (Transport Canada Risk Assessment). A brief description of the approach and key findings of this assessment is provided in Chapter 7. In this chapter, uncertainties associated with that risk assessment are identified and discussed in detail because of its potential influence on future policy, spill preparedness and response activities.

8.13.2 Risk assessment methods

8.13.2.1 Spill frequency

The TCRA considered production (offshore wells) and transportation sources and divided spills into four sizes: 10-99.9 m^3; 100-999.99 m^3; 1000-9999.9 m^3 and >10,000 m^3.

Each of the four sectors was divided into subsectors that were further subdivided into three zones representative of near-shore, intermediate and deep sea environments. A total of 77 zones were allocated a frequency of spill and an environmental sensitivity, which were then applied to generate a risk est imate (WSP and SL Ross 2014a).

Estimated spill frequency was based on the past 10 years of spill data from Canadian and international data sources. Spill fre quency was estimated for crude, refined cargo and fuel according to the relative volumes transported across each subsector. The Panel focused on the results for crude oil.

Uncertainty Regarding Spill Frequency

- The TCRA noted that while the variability in spill rates is not wide among different governance structures around the world except perhaps at the highest end of spill size ranges, the use of worldwide incident data may have overestimated the likelihood of spills in Canada given the robust marine governance regime and the actual spill record. This assumption is subject to debate based on evidence such as the global review of environmental damage liability regimes by Goldsmith et al. (2014). Furthermore, desp ite advances in technology and improved safety protocols, accidental releases, such as the recent *Marathassa* spill, will continue to occur (Section 8.12). The effects of increases in the volume of tanker traffic on spill rates in the various sectors and subsectors were not discussed in the report.
- Probability estimates by the TCRA for the St. Lawrence Seaway were identified to be conservative as they were not corrected to account for closure of the Seaway about 30% of the year due to ice cover. However, their analysis did not include US traffic in the Great Lakes , which would have provided a similar precautionary approach for evaluating the risk of spills in the Great Lakes.

8.13.2.2 Environmental sensitivity

The TCRA estimated environmental sensitivity of each zo ne using three indicators: physical sensitivity (PSI), biological resource (BRI), and human -use resource (HRI). The environmental sensitivity index (ESI) was calculated as follows: ESI = 0.3(PSI) + 0.5(BRI) + 0.2(HRI). The weights applied to each indicator were from a review of costs, such as the influence of physical factors in determining cleanup costs.

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The PSI rank was related to zone sensitivity, natural oil persistence and ease of cleanup.

- Exposed rocky shores were ranked least sensitive;
- Most sensitiv e were sheltered rocky shores, tidal flats, marshes, wetlands, eelgrass and ice infested waters; and
- Gravel was ranked in the middle.

A 0.3 PSI for open sea was based on average cost of cleanup per tonne compared to near-shore environments The BRI and HRI were based on a method developed for Australia (DNV 2011 cited in WSP and SL Ross 2014a). It was assumed that the BRI of intermediate and deep sea zones that did not include any of the identified biological components (e.g., species at risk, birds, mammals, reptiles, fish, meroplankton and invertebrates) would be related to that of the nearest near-shore zone times a weighting factor of 0.4x for intermediate and 0.1x for deep sea. Weights attributed to each biological component included data from the Strategic Environmental Assessment for the Gulf of St Lawrence study (GENIVAR 2013 cited in WSP and SL Ross 2014a) and from other marine environmenta l studies (dredging, harbour, etc.).

The HRI was based on a combination of commercial fishing intensity, tourism employment intensity, freshwater use intensity and freight tonnage index. Weights applied to each component were: 0.55 for commercial fishing, 0.2 for tourism, 0.15 for water usage and 0.1 for port industry. Commercial fishing intensity scores did not include recreational or traditional fishing due to absence of comparable data, but the authors maintained that the index would provide a sufficient indicator of overall vulnerability. Tourism Employment Intensity was based on the ratio of tourism industry employment versus total employment. The Freight Tonnage Index was based upon total tonnage in each port. The use of fresh water by humans (Freshwat er Use Intensity) was based upon coastal population as a proxy because data on water intakes were not available.

<u>Recommendation:</u> Future relative risk assessments should include explicit consideration of intermediate and deep sea zones with greater sensitiv ity, particularly in zones with current or projected offshore oil production.

The TCRA report noted that the environmental sensitivity values were generally lower for intermediate and deep sea zones. They noted exceptions due to the availability of data fo r some particular, sensitive intermediate and deep sea zones. However, there was no attempt to include these areas in the analysis (for example, as exception layers). A discounting term may be appropriate in order to account for increased capacity for dilution, dispersion and biodegradation in some of the zones or subzones.

<u>Recommendation:</u> Future relative risk assessments should build upon the TCRA results by focussing on high-sensitivity areas within each of the assessment zones in order that preparedness and response plans can include explicit plans for the areas with the highest potential consequences.

The TCRA report presented ESIs as average values for an entire zone. The authors noted the shortcomings of this approach as the use of averaged numbers res ults in loss of detail —high sensitivity areas within a zone might be concealed if surrounded by relatively low -sensitivity areas due to the subsectors being so large.

<u>Recommendation:</u> Consideration should be given to the refinement of the PSI values based o n recent shoreline mapping data (an ongoing initiative of Environment Canada) and site-specific data from past spills.

<u>Recommendation:</u> The methods for development of BRIs require review and refinement in order to increase the reliability of the relative ri sk indices at the subzone level. These refinements should

reflect the nature of the receiving environment within subzones and the specific valued components within the subzones.

<u>Recommendation:</u> The TCRA should be updated to include a fresh water-specific set of ESIs. Such an assessment would be relevant not only to ship-source spills, but to spills from pipelines, crude-by-rail and truck transport.

The reliability of the BRIs depended on the quantity and quality of data on the components (e.g., birds, mammals, fish, etc.) in terms of where they are located within the zones plus the weighting for sensitivity. Data for some components in some of the subzones were limited or non-existent. The information presented in the TCRA on chronic effects provides evidence that raises questions about the weighting of sensitivity; e.g., Table 4.1 in the TCRA. For example, fish are given a medium-low sensitivity despite the sensitivity of early life stages to PAHs, as demonstrated in laboratory as well as field studies (Chapter 4). However, evidence from spills around the world seems to indicate that fish populations are not as sensitive as other marine biota. The cumulative impacts of stressors that can exacerbate the effects of spills (e.g. increasing water temperature and/or nutrient loads) were not considered under the BRI. Within the TCRA report there seems to be marine bias in the weighting approach used for sensitivity (probably due to the larger database available); thus, the applicability to fresh water (i.e., the St. Lawrence River and Great Lakes) is questionable.

<u>Recommendation:</u> Future relative risk assessments should include refined human use resource indices, which include the uses of waterbodies for traditional purposes by Indigenous peoples, and which include the presence of critical municipal, agricultural and industrial uses that may be adversely affected by a spill. The Panel notes that the TCRA report includes a similar recommendation.

The appropriateness and completeness of the HRI is subject to criticism since the inputs to the index, and the weightings applied to those inputs, would be expected to vary significantly within zones and subzones. For example, the presence of critical water intakes for agriculture in areas with lower human population size along the St. Lawrence River might justify an increased weight for water use intensity in those areas. In addition, the HRI did not incorporate the significance of traditional resource uses by Indigenous peoples.

8.13.3 Results of the risk assessment

The results of the TCRA are described in Chapter 7, whereas this chapter focuses on key parameters determining the outcome of the risk assessment, followed by a review and discussion of uncertainties associated with the TCRA results.

WSP and SL Ross (2014a) identified the key parameters influencing the relative risk outcomes as:

- Oil persistence, a consequence of chemical composition and microbiological sensitivity or recalcitrance.
- Extent of the spill over time.
- Potential movement of the spill.
- Characteristics of the receiving environment.
    - Shoreline habitat, sediment type, topography, currents, hydrology;
    - Presence and concentrations of factors that would enhance natural attenuation or biodegradation, such as nutrients, electron acceptors, etc.;
    - Coastal resources in area of influence of spill;
    - Physiological and behavioural characteristics of coastal resources; and
    - Type and intensity of human activities.

- Type and volume of the spill.
    - Toxicity, bioaccumulation rates;
    - Volume and exposure concentrations in various media (water, sediments, soil, food, air);
    - Direct versus indirect exposure via food;
    - Exposure duration; and
    - Time of year—organism life cycles plus weathering.
- Type and effectiveness of the cleanup response.
    - Possibility of damage from cleanup activities.

The TCRA authors stated that effects on natural habitats in fresh water will resemble those of marine spills. However, they also stated that spills in fresh water have a much greater potential for contaminating potable water supplies, affecting areas of concentrated populations, damaging manmade structures and disrupting other human activities such as recreation.

The TCRA authors also noted that, in most habitats, site recovery from an oil spill will occur within 2-10 years, and cautioned that perception of contamination can last longer than actual contamination, with economic effects on fisheries and tourism sectors. The perennial and thorny question "How clean is clean?" has been noted in Chapters 6 and 7.

It cannot be assumed that the TCRA risk assessments are uniformly precautionary. The TCRA assessment is conservative in that it did not assume any marine oil spill prevention measures or responses, such as mechanical recovery or dispersant application. However, the overall level of conservatism in the assessment is difficult to determine because components within the assessment, such as environmental sensitivity, may not have been uniformly conservative.

<u>Recommendation:</u> Preparedness planning should consider information from Indigenous, provincial, territorial and municipal sources in order that smaller-scale, higher-risk locations can be identified. For example, the Strait of Georgia and the Gulf of St. Lawrence have areas of concentrated seabird and migratory bird use that would be a high priority for specific response measures targeted at reducing impacts to birds. Many of these areas have already been mapped at a finer scale than presented in the TCRA. Provincial laws and regulations must be considered in preparedness to ensure harmonization and avoidance of redundancy.

Not including data from provincial, territorial, and municipal sources is understandable for such a high-level assessment; but it produced significant limitations in the risk analysis.

Except for the St Lawrence River, the assessment did not include rivers such as the Mackenzie River and Fraser River, which have barge traffic and could have increased traffic in the future.

<u>Recommendation:</u> First Nation, Inuit, and Métis jurisdictions must also be considered and included in preparedness planning and broad consultation with Indigenous peoples should be part of such planning.

Any and all future preparedness planning should explicitly include First Nations, Inuit, and Métis jurisdictions as well as broad consultation with all Indigenous peoples. Indigenous peoples are often the most knowledgeable groups regarding local and regional conditions which would affect preparedness planning. Indigenous peoples are also often the closest people to spills in remote locations; therefore, they can be well-placed to be among the first responders (if capacity exists in Indigenous communities).

<u>Recommendation:</u> Any future relative risk assessments should include weather as a factor within the risk scenarios.

It was logical that local or regional -scale weather data were not included in the TCRA high -level assessment. However, consideration of the relative frequency of extreme weather events among the zones used in the assessment would have added value since the weather can be an important determinant of consequences (e.g., due to delays in response, weather effects on effi cacy of response measures and weather effects on dispersion of the oil).

Recommendation: The TCRA should be revised to include risks of spills from U.S. -bound tanker traffic in Canadian waters.

Risk in the Pacific sector was related largely to tankers going to refineries in Washington State, and the authors stated that, therefore, these spills would not be subject to Canada's spill response regime, despite close proximity to southern British Columbia's coastal areas. There is also concern that the risks of spills from U.S.-bound tanker traffic within Canadian waters was not considered.

Recommendation: The Panel has made several recommendations with respect to research on the fate and effects of diluted bitumens. Results of this research should be include d in future risk assessments, particularly for scenarios where research indicates that the fate or toxicity of diluted bitumens would differ significantly from conventional crude oil.

The modeling of the behaviour of spills did not distinguish diluted bitumens from conventional crude, and it was not clear whether critical differences between marine and freshwater systems with respect to the processes were included in the model (drifting, spreading, evaporation, photooxidation, natural dispersion-dissolution of oil in water, water -in-oil emulsification, biodegradation and sedimentation; Chapter 2).

Recommendation: The current lack of inland shoreline sensitivity information in the open literature is a serious gap affecting the ability to apply risk -based prep aredness and response planning. Prioritization of areas requiring mapping could be based upon evaluation of the intensity of human use, the current knowledge of the relative sensitivity of ecosystems, and the availability of information from various source s. For example, high -priority areas for mapping might include areas upstream and downstream of hydroelectric dams.

Shoreline mapping is still incomplete, particularly inland, including along the Great Lakes. The Panel is aware that industry is doing some mapping and that Environment Canada continues to work on mapping marine coastal areas. The Panel also understands th at Environment Canada has been working with Alberta, British Columbia, and CN Railway on mapping of rail corridors and that similar work was done in Quebec (S. LeBlanc, pers. comm). Environment Canada has been approached by Canadian Pacific regarding conducting similar work along their railway corridors (S. LeBlanc, pers. comm.).

Recommendation: National-scale relative risk assessments used for preparedness and response planning must be based upon reliable and consistent data and analysis . Therefore, standardized methodology for environmental sensitivity mapping is an urgent requirement.

Apparently, even when environmental sensitivity data are available, they are obtained using different methods. For example, according to the TCRA, survey methods used by eac h regional department of the DFO may differ, and bird data are obtained using different types of datasets, resulting in regional disparities.

Recommendation: Future refinements to relative risk assessment must include explicit modeling of freshwater systems, taking into account the unique features of freshwater environments tha t affect exposure and effects.

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The TCRA assumption that effects in fresh water will resemble marine spills is problematic, as illustrated by the case studies reviewed earlier in this chapter and the review in Chapter 4.

Recommendation: The Panel recommends that the relative risk assessment method could be applied at a smaller scale for the purposes of identifying the critical areas for protection and preparedness, and could be used to identify critical data gaps.

As the authors state, rare events are difficult to predict —especially the really large spills . Therefore, preparedness planning should focus on events where consequences would be highest.

Recommendation: The Panel strongly recommends the development of a national, consensus-based set of indicator species for each of Canada's high -risk major marine offshore and inshore zones. Once these indicator species have been selected, coordinated research programs among academia, industry and government should be developed that: a) compile existing information; b) identify and prioritize critical data gaps; and c) conduct research to fill the priority data gaps. (See recommendations in the Section 8.15 for additional details.)

Recommendation: The Panel recommends that future national or regional -scale relative risk assessments should include indices of current status of receiving environments, with emphasis on existing levels of anthropogenic stressors. Future risk assessment could also combine risks of tanker spills with risks of pipeline, rail and road spills for a cumulative oil spill risk in the most se nsitive receiving environments.

The overall TCRA environmental risk index appeared not to address cumulative risks or the existing state of the environment (which may already be degraded and at increased risk; reviewed in Chapter 4).

8.13.4 Major Conclusions and Recommendations from the Transport Canada Risk Assessment Report

Some of the key conclusions and recommendations presented in the TCRA report were:

- Spills in the 10 to 1000 m^3 range generate higher overall risks both respect to frequency and consequence;
- Eight of 10 highest risk areas are near-shore and two of the 10 are intermediate —this conclusion has implications for response capabilities and preparedness;
- The Marine Pollution Incident Reporting System database on pollution incidents maintained by the Canadian Coast Guard should be examined for improvements regarding comprehensiveness and quality of data;
- In future risk assessm ents, consideration could be given to effects of spill response according to oil type and spill size, as well as remoteness, weather and other factors;
- All regions should attain the same level of detail regarding environmental data, and these data should be in GIS (geographic information system) format;
- Regarding physical sensitivity, more information on evolution of coastal areas facing climate change impacts, in particular for sensitive components such as coastal wetlands, is urgently required. I ntegrating data on littoral geomorphology to capture erosion impacts would be a valuable refinement– these data are often available at the provincial level;
- Regarding biological resources, data should be incorporated for all protected marine areas under various jurisdictions and Ecologically and Biologically Significant Area (EBSA) or equivalent layers, and risk analysis for fresh water should be considered separately from marine analyses due to differences in the type of data available; and
- Regarding human resource use, information should be incorporated for provincial and municipal conservation/protected areas, for freshwater intake and utilization (drinking water, agricultural

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and industrial) and for archeological and cultural heritage sites. The current risk estimates should be overlaid with information on Aboriginal communities.

The Panel supports the recommendations of the TCRA report.

8.14 Risk Assessment for Marine Spills in Canadian Waters North of 60th Parallel

A relative risk assessment for oil spills in Arctic waters was conducted, but only refined products were considered because crude oil is not currently transported in the Arctic (WSP and SL Ross 2014b). Therefore, the Panel reviewed this Arctic Marine Risk Assessment (AMRA) report as a general indicator of risk of oil spills in the Arctic, with emphasis on drivers of risk and uncertainties within the risk calculations.

The Panel's review focuses on the environmental risk index component of the AMRA since calculation of spill frequencies appeared to be the best possible given the available data. The authors discussed the overall poor coverage of Arctic navigational charts (except for the Beaufort Sea and Foxe Basin) but decided not to apply increased weight to spill frequency due to poor navigational information in specific subsectors. The reason is that there had not been a significant number of reported spill incidents related to navigational issues and, for the most part, marine traffic in the Arctic is performed by long-time operators who use up-to-date charts and are familiar with their routes.

The logic behind the use of the same basic methods for calculating risk for North of 60 as for South of 60 is debatable. The PSI represents the degree of difficulty involved in coastal cleanup due to factors such as remoteness and/or distance from first-response centres. Therefore, the weighting attached to the PSI for Arctic conditions arguably should have been higher than for the south. Furthermore, the HRI might need upward adjustment due to use by Indigenous peoples.

<u>Recommendation:</u> Shoreline mapping should be conducted in areas where oil and gas exploration and development is already occurring or is expected, e.g., Beaufort Sea and the Mackenzie Delta. The mapping should also consider other human activities that would affect relative risk, such as mining, where large port facilities have been proposed involving not only transport of the minerals out but fuel in, and increased commercial shipping traffic through the Northwest Passage, which brings greater risk of fuel oil spills. The current rapid rates of coastal erosion should also be noted and mapped for the purposes of continued monitoring.

The AMRA authors assumed that ice-free shorelines are the most sensitive because the presence of ice prevents the oil from reaching the shoreline. This was a simplistic assumption as there could be times when the shoreline is partially ice-covered and the situation will be very dynamic. The highest risk would probably be during the seasonal changes when multiple processes might control fate and effects (e.g. erosion, scouring, slumping that would take oil and mix it in with shoreline materials or near-shore sediment materials).

The AMRA authors noted the lack of information regarding Arctic shoreline types.

<u>Recommendation:</u> The Panel acknowledges that shipping traffic volume is low in the Arctic. However, shipments of fuel oil to Arctic coastal communities do occur, and the risks of spills of fuel oil should be assessed. Furthermore, in preparation for increases in Arctic traffic the AMRA methodology needs to be revised and tailored for the Arctic.

Risks of spills during shipment of fuel oil by ship and barge as well as transfer of fuel from barges to storage facilities require assessment in order to inform preparedness planning. The methods used for the AMRA biological resources index were basically the same as for the South of 60° TCRA study. The Panel questions the use of the same methods for the Arctic as for more southerly zones, particularly

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sensitivity weights. Sensitivity in the Arctic should consider more than marine species and habitats, as river deltas and the rivers themselves are important for anadromous fish [7] and other sensitive uses (including human uses).

The calculation of the HRI involved the following:

- Coastal population index (CPI);
- Tourism index (TI);
- International freight tonnage index (IFTI);
- National freight tonnage index (NFTI); with
- HRI = 0.7(CPI) +0.2(TI)+0.05(IFTI)+ 0.05(NFTI)

The AMRA authors state that, "since the majority of the population in the area of study is Inuit, the coastal population indicator captures explicitly the human use for these communities."

This assumption that the coastal population captures human use of the coastline is an over-simplification because many coastline areas with zero population can have high traditional use value. The Panel questions whether the CPI would be found to be sufficient by Inuit people.

<u>Recommendation:</u> Additional research should be conducted in the near future to capture both increases in marine traffic and the activities of commercial fisheries in the regional risk assessment.

The AMRA report acknowledged the value of royalties paid to Nunavut by offshore turbot and shrimp fishing companies and also acknowledged the value of Arctic char and sealing to communities and state that these activities are vulnerable to marine spills. It was felt that the large scale of the assessment limited the incorporation of commercial fisheries as their activities may not be associated with coastal environments.

<u>Recommendation:</u> Current environmental risk index (ERI) values are low or very low because of projected low spill frequency. Therefore, the Panel is uncertain whether ERIs, in their present form, are useful for planning. The Panel suggests using the ESI component of the ERI equation as a means of identifying priorities. However, the way that the ESIs are calculated would have to be revisited using input from communities and scientists because the Panel is not convinced that the current ESI methods are relevant and representative of Arctic conditions.

The authors state that, from an emergency planning point of view, the ERI can be interpreted as a relative measure of the importance of risks associated with oil spills. A large ERI value implies a relatively higher risk of economic and environmental damage. Therefore, the index could be applied to damage reduction in Arctic waters if combined with emergency planning by authorities. However, the results of the ARMA did not provide a particularly useful level of resolution, given the scale of the Arctic and the location of population centres that may serve as the location for Arctic preparedness.

The AMRA report gave little consideration to risks in the Mackenzie River and Great Slave Lake (and other freshwater areas), except with respect to the Mackenzie River delta where it enters the Beaufort Sea. Traffic is low in these areas; however, the same can be said for the Beaufort Sea and the southern part of Hudson Bay, including James Bay and the Foxe Basin, where ERIs were high in some cases. Given the description of the ecology of the Mackenzie River and Great Slave Lake in Section 4 of the AMRA report, and the presence of EBSAs, it is puzzling that the calculated risks for these two areas did not exceed the "relatively low" category. There were no comments about the future risks for the River or the Lake in Appendix 2 of the AMRA report.

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[7] *Anadromous fish hatch in fresh water, spend most of their life in the sea, and return to fresh water to spawn; e.g., salmon.*

8.14.1 Recommendations from the Arctic Marine Risk Assessment Report

The Panel concurs with the following recommendations made in the AMRA report:

- Matching vessel codes from the Automatic Identification System database and Transport Canada commodity database would allow consideration of the exact trajectory of specific cargo —this should be considered for the Arctic as well as other regions;
- The current analysis is based on yearly estimates of spill frequencies. The estimate could be refined by taking into account the length of the shipping season within each subsector and thereby considering seasonal variations that influence risk estimates;
- Evaluation of coastal areas facing climate change impacts is required;
- A separate analysis for fresh water (Mackenzie River an d Great Slave Lake) is necessary to account for differences in critical processes affecting fate and transport of oil as well as effects; and
- Commercial fisheries information should be incorporated into the HRI.

Recommendation: The Panel also recommends th at future relative risk assessments for the Canadian Arctic include spills from exploration and production in adjacent U.S. waters of the Beaufort Sea as well as future drilling in Canadian waters.

8.15 Assessment of Marine Oil Spill Risk and Environmental Vuln erability for the State of Alaska

The Panel reviewed an assessment of oil spill risk for Alaska (Reich et al. 2014) to compare and contrast the methods used with those used for the two Canadian risk assessments reviewed above. The results of the Alaska risk assessment are briefly described in Chapter 7.

The Alaska assessment was at a smaller spatial scale than the Canadian AMRA assessment, but still covered a large area. This scale required the use of similar generalizations regarding spill behaviour and effects as those used in the AMRA.

Recommendations based on the differences between the Alaska and Canadian (AMRA) assessments that should be considered for future risk assessments in Canada are presented below.

Recommendation: Future relative risk assessments conducted for the Canadian Arctic include season-specific influences on sensitivity.

The Alaska assessment included season-specific vulnerability, which is very important in Arctic regions.

Recommendation: Future risk assessments for the Canadian Arctic should include consideration of crude oils, including exploration and oil production facilities.

The Alaska assessment considered crude oils, heavy oils, light oils and distillates , whereas the Canadian assessment focused on refined products only.

Recommendation: The Panel considers the inclusion of dispersion modeling to be an essential component of risk assessments of oil spills, particularly at the regional scale. However, the Panel notes that the dispersion modeling done in support of the Cana dian assessment was basic and would require refinement for application to finer scales (with the accompanying requirement for additional data including oil interactions with ice, and biodegradation potential as bioremediation may be the most feasible cleanup strategy in remote Arctic areas).

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Weathering processes and the amount of oil reaching shore were not modeled in the Alaska assessment . However, the report recommended that for regions with high relative risk, trajectory and fate modeling should be conducted.

Recommendation: The Panel notes that the vulnerability method used in the Alaska assessment is applied at a finer -scale and does not rely on broad sensitivity categories applied to entire groups; therefore, this approach may be more appropriate if fi ner-scaled relative risk assess ments are conducted in Canada.

Vulnerability was assessed for Alaska receiving environments using habitat (shoreline, bottom marine and sea ice) and species sensitivity and recovery information compared with "sensitivity" in the Canadian assessments, which combined physical, biological and human-use indices.

Recommendation: The Panel suggests that an assessment of future risks in the Canadian Arctic include a critical review of the preparedness and response/mitigation options applicable to Arctic conditions and the likelihood of these measures being effective in reducing both the probability and consequences of oil spills, with special consideration of spill site remoteness and distanc e from first-responder centres.

The Alaska assessment assumed that risk mitigation would reduce the probability of an incident becoming a spill event and also assumed increased effectiveness of spill prevention and risk mitigati on measures to reduce spillage.

Recommendation: The Panel suggests that a Spill Risk Calculator tool for Canada would be beneficial to preparedness planners and responders.

The authors of the Alaska study provided a Spill Risk Calculator Tool and noted that the transparency of the method allows for quick updates of results as more data become available.

The Alaska assessment method was very data -intensive. A similar quantity of relevant data would not be available for the Canadian Arctic except for a few relatively small areas.

Recommendation: The Panel strongly recommends a national database of information required for relative risk assessment in the Arctic . T his database would include government, academic and industry sources, as well as traditional knowledge sources.

Recommendation: The Panel suggests that a more detailed as sessment of the Beaufort Sea be a priority for the next round of relative risk assessments. The assessment could be used to identify critical data gaps for parameters that drive risk in the Beaufort Sea.

Both the Alaska and Canadian assessments identified the Beaufort Sea as a high relative risk area for crude oil spills in the future.

Recommendation: Although it would require some educated guesses (preferably in collaboration with industry), incorporation of cleanup costs would add value to the results of relative r isk assessments for the Arctic.

Neither the Alaska nor the Canadian assessments included cost of cleanup because information on such costs was lacking (due to the fortunate lack of major spills).

In summary, the Alaska analysis provide d a conserv ative relative environmental risk map based on maximum most probable discharge or worst -case discharge. The assessment did not account for weathering and did not model amounts reaching shorelines, but it did include seasonal differences, a

fairly large num ber of discrete habitat types, and a cross -section of species or species groups that are assessed for vulnerability using several parameters. The Alaska analysis did not provide socioeconomic analysis, which can be regarded as a significant weakness.

8.16 Assessment of Risks of Hypothetical Spills from the Proposed Northern Gateway Pipeline

Detailed ecological risk assessments were conducted to evaluate risks to freshwater and estuarine resources from hypothetical spills along the proposed Northern Gateway pipeline (NGP) route (Green et al. 2015). The risk assessments conducted for the NGP were quantitative and detailed, in contrast to the relative risk assessments reviewed above. The purpos e of the NGP assessment was to evaluate the risks associated with a full bore pipeline rupture at discrete locations.

Ecological risk assessment (Figure 8.13) is conducted using a standard framework that starts with Problem Formulation, proceeds through exposure and effects analysis and concludes with characterization of the risk. Problem formulation includes: development of risk management goals and objectives; identification of assessment and measurement endpoints; determination of the spatial and temporal context; screening for stressors of concern; and selection of valued ecosystem components (VECs). Engagement of interested parties in problem formulation can reduce the level of criticism and debate because the range of values can be represented via selection of goals, endpoints, receptors, etc. Exposure assessment is conducted via measurement and/or modeling. Effects assessment is usually conducted by comparing predicted or measured exposures with derived effects thresholds based on toxicity test results. Risk characterization estimates the level of effect, most commonly through the calculation of so-called hazard quotients (the ratio of the predicted exposure and the effects threshold).

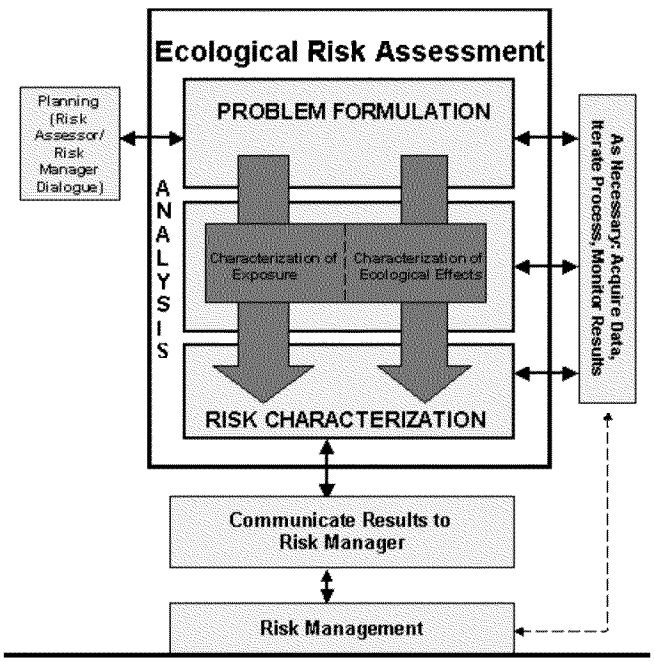

Figure 8.13 Framework for Ecological Risk Assessment. *Image from EPA (1998).*

These detailed assessments included modeling of dispersion and resulting concentrations of whole oil and oil components on the water surface, shorelines, in the water column and in sediments of four different receiving streams that were selected on the basis of environmental, socioeconomic, cultural and

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traditional values (Green et al. 2015). The receiving streams also reflected a range of watercourse types and had limited access, thus increasing response times. The predicted concentrations in and on water, in sediments and along river and estuary shorelines were then assessed against thresholds for acute and chronic effects for a range of species.

The four locations selected for the NGP assessment were:

- Chickadee Creek – low gradient interior river tributary discharging to the Athabasca system upstream of a populated centre in Alberta (Whitecourt);
- Crooked River – low gradient interior river with wetlands entering a lake system in the Interior Plateau, BC;
- Morice River – high gradient river along the western boundary of the Interior Plateau with sensitive fisheries resources and downstream human population; and
- Kitimat River near Hunter Creek – high gradient coastal tributary discharging to a large watercourse with sensitive fisheries resources, downstream human population and discharging to Kitimat River estuary (Green et al. 2015).

The Panel's review of the NGP assessment focuses on key findings of the dispersion modeling and toxicity assessment and compares these findings with the spill case studies presented earlier in this chapter. However, before presenting our comments on the detailed findings, we have some comments regarding important aspects of the spill scenario.

The spill scenario was a so-called 'full bore rupture' releasing a volume of between 974 to 3,321 m^3, which would place the releases into the "moderate" or "large" spill categories used in the relative risk assessments conducted for tanker spills by WSP and SL Ross (2014a,b). The hypothetical volumes released depended on the volume of oil that would be contained within the pipeline from the full-bore rupture point to the nearest shutoff valves in the pipeline on either side of the rupture. It was assumed that 13 minutes would elapse before shutoff valves were activated. "While spill durations would likely be longer, as oil would drain from the pipeline, all oil was released in 13 minutes to represent conservatively high in-water concentrations" (Horn and French-McCay 2015).

The NGP risk assessment did not assume any efforts to prevent oil from entering the river or migrating downstream, nor did it assume any recovery of the oil. This 'worse-case scenario' for the conditions outlined maximized impacts for those conditions; however, the overall degree of conservatism in the assessment is subject to discussion and debate.

8.16.1 Review and Recommendations Based on the Key Findings of the Northern Gateway Pipeline Ecological Risk Assessments

<u>Recommendation:</u> Risk assessment scenarios that assume a more likely combination of factors (less than full-bore rupture with a longer elapsed time to response) would be more relevant and comparable to actual spill incidents.

In the Panel's view, the assumed time period to shutoff is not conservative, given recent experience; e.g. the 17-hour delay for stopping the Kalamazoo spill (Section 8.10) and an unknown period of time (but possibly several days) for the Nexen emulsion spill (Section 8.13). The assumed 13-minute elapsed time before shutoff of oil flow significantly constrains the spill volume, thus fundamentally affecting all subsequent outputs from the risk assessment.

<u>Recommendation:</u> Detailed risk assessments conducted in support of applications for new projects, such as pipelines and production facilities, should include winter conditions, including full or partial ice cover and spring flood conditions with accompanying ice jams.

The NGP modeling did not consider the presence of ice on or in the receiving streams. It is likely that the presence of ice would be a 'risk driver'. Oil might flow on top of the ice to impact river shorelines above the normal flow height. Heated dilbit might locally melt ice and thereafter travel below the ice where it would be difficult to detect and intercept. The presence of snow and ice could decrease evaporation — alleviating explosion hazards but increasing toxicity risk at spring melt. Any assessment of ecological risks in Canada should consider seasonal differences in parameters contributing to fate and behavi our as well as to toxicity.

Recommendation: Future detailed risk assessments should consider the influence of animal behaviour on exposure to oil spills.

Animal behaviour was not considered in NGP exposure modeling. As illustrated by some of the case studies reviewed above, fidelity to particular habitats can be an important contributor to animal exposure, especially to oil deposite d to sediments or stranded on vegetation or organic debris. It would have been interesting to compare risks to receptors such as salmonids with and without assumptions of fidelity to particular locations in the receiving streams (e.g., spawning sites). British Columbia coastal rivers attract grizzly bears during salmon spawning migrations, providing another example of a behaviour -related risk scenario.

Recommendation: Differences and similarities between modelled and observed fate and transport of dilbit, synthetic oil and condensate should be used to develop future research projects as well as guidance for standardized monitoring of oil spills.

Several results of modeling of the fate and transport of three products —dilbit, synthetic oil and condensate—were consistent with observations during actual spill events (Horn and French -McCay 2015). The dilbit selected was the dense and highly viscous MacKay Heavy Bitumen diluted with Synthetic Light Oil. The synthetic oil was an intermediate Syncrude Synthetic Light Oil. The condensate was a highly aromatic and low viscosity CRW Condensate Blend (Table 2.2). The following list presents the NGP predictions and the corresponding observations from case studies (in italics):

- Extensive shoreline oiling within confined reaches of small creeks and parts of the larger rivers was predicted, *as was observed in the Pine River (light crude) and Kalamazoo River (dilbit) spills. However, there was little consideration of the interaction between river stage, exposed shoreline length and amount of shoreline oiled on falling or rising flow stages;*
- The magnitude of river current during the spill event was of primary importance —higher flows resulted in more extensive oiling of shoreline and transport further downstream transport, *as was observed in the Pine River and Kalamazoo River spills;*
- Higher wind cases for a hypothetical dilbit spill resulted in oil deposition to sediments as a result of wind preventing entrained oil from resurfacing an d allowing more time for interaction with suspended sediments and eventual sinking. *The Wabamun Lake spill case, although it was a lake rather than a river, also illustrated the importance of wind in dispersing and mixing heavy crude in the water column as well as stranding heavy crude along shorelines;*
- Hypothetical dilbit spill cases typically resulted in the most extensive oiling of shorelines compared to the other oils, *as was observed in the Kalamazoo spill; however, the Pine River spill of light crude also resulted in extensive oiling of backwaters and side channels; and*
- Condensate was predicted to stay buoyant with little or no deposition –slicks were maintained longer under low flow and low wind; and extensive evaporation was predicted. *These predictions mirror observations of the diluent in the Kalamazoo spill.*

Other predictions did not have analogous spill cases, either because of the type of oil or because of the type of receiving environment. These predictions included:

- Vertical turbulence would be induced by bottom roughness of the stream bed and, secondarily, winds would entrain oil into the water, modified by the oil viscosity. *This was not observed in the river case studies, although there were no explicit attempts to examine entrainment in the Pine River case;*
- Floating dilbit that was not entrained or stranded ashore was predicted to emulsify quickly to form highly viscous mousse, which prevented additional entrainment and sinking as it was carried farther downstream. *Dilbit mousse formation was not reported for the Kalamazoo spill, nor has it been observed in open -tank experimental spills (Chapter 2). Dilbit is not as amenable to emulsification or mousse formation as are conventional crude oils;*
- Dissolution of soluble and sparingly soluble aromatics into the water was predicted to be enhanced by entrainment as small droplets. *This particular mechanism was not studied in the Pine River case and there are no published data regarding this mechanism for the Kalamazoo spill s. H owever, it was suggested that wind -induced formation of Langmuir troughs in Wabamun Lake might have contributed to vertical mixing of the spilled heavy fuel oil;*
- Under high flow and high wind conditions, bottom roughness and wind -induced mixing was predicted to keep large amounts of condensate entrained in the water column. *The predominant fate of diluent (Kalamazoo River) and lighter fractions of light crude (Pine River) was rapid surface dispersion and evaporation; however, temperature and current regimes in the NGP assessment were very different;*
- Above a certain site -specific threshold, dependent on streambed roughness and total suspended solids, higher winds were predicted to maintain the entrained oil in the water column allowing large amounts of de position to the sediment. *The mechanisms producing deposition to the sediment were different in Pine River (where the primary drivers were slow current and fine -grained sediments) and the Kalamazoo River (where the primary drivers were factors leading to the production of OPAs); and*
- The predicted behaviour of synthetic oil was distinctive and contributed to higher risks being associated with this type of oil. The predicted behaviour of synthetic oil included:
  - A relatively large percentage that became entrained and potentially settled into sediments. Synthetic oil under both high flow and low flow conditions resulted in the largest amounts of oil deposited to sediments;
  - The entrained oil and high concentrations of dissolved aromatics were in many cases transported down the full length of the modeled river; and
  - As a result aquatic biota were exposed to acutely toxic concentrations along the modeled reaches.

<u>Recommendation:</u> The Panel recommends that post-spill monitoring guidance include protocols for examining hyporheic flow.

The Panel notes that the NGP assessment did not include a detailed consideration of entrainment of dissolved and particulate oil into coarse sediments by downward hyporheic flows. These flows characterize salmonid rivers and may expla in the fate of much of the unaccounted for oil in spill case studies. The NGP risk assessment mentions hyporheic flows only in passing as an inflow of groundwater that would act as a diluent of any dissolved oil in gravel beds. To date, the Panel is not aw are of any reports of post-spill systematic investigations of oil distribution in stream gravel, particularly as it relates to hyporheic flow down -welling zones. The same principles apply at the river's edge where lateral hyporheic flows generated by river bends transfer river water (and potentially oil) into ground water of riparian zones.

<u>Recommendation:</u> Differences and similarities between modeled and observed acute and chronic toxicity of dilbit, synthetic oil and condensate should be used to develop fu ture research projects as well as guidance for standardized monitoring of oil spills.

Predicted acute toxicity to water column organisms was greater for synthetic oil and condensate than for dilbit because of the high viscosity of dilbit (which impeded ent rainment), the lower concentrations of low molecular weight compounds and PAH s in dilbit, and because of the smaller water areas and river lengths affected by dilbit. However, acute injury to wildlife and emergent vegetation was usually highest in modeled dilbit cases due to higher viscosity that kept it floating on the surface or adhering to the shorelines.

These predictions are consistent with the spill case studies reviewed by the Panel. *Acute toxicity to aquatic life appeared to be minimal during the Kalamazoo spill, with no reported fish deaths. In contrast, thousands of fish were killed due to the spill of light crude into the Pine River. Acute injury to wildlife was low during the Kalamazoo River spill. Death of emergent vegetation was observed duri ng the DWH spill, but not during any of the freshwater spill cases reviewed.*

Assessment of the potential for chronic toxicity extended for up to one or two years post-spill. The chronic NGP assessment addressed: weathering on shoreline soils and in stream sediments; partitioning of hydrocarbons from sediment back into pore water in streambed spawning gravels; potential effects to fish eggs and embryos present in spawning habitat; and chronic risk to receptors that are higher level consumers (Stephenson et al. 2015). Two models were used to evaluate chronic fate and bioavailability: these models predicted the fate as a mixture of 61 hydrocarbons, including the Canada -wide Standard aliphatic and aromatic TPH fractions, BTEX and other monoaromatic hydrocarbons, PAHs and other miscellaneous hydrocarbon compounds. Long-term exposure to a full suite of alkylated and non-alkylated PAHs that may be dissolved in the interstitial water of stream bed gravels was considered for potential effects on developing fish eggs a nd larvae. Chronic exposure point concentrations were calculated for water, sediment, oil, vegetation and fish tissue at time points of four weeks and one to two years (Stephenson et al. 2015).

Chronic effects benchmarks were developed from the Target Lipid Model (TLM) of Di Toro et al. (2000) and Di Toro and McGrath (2000). Canada-wide standards for PAHs in soils were used for the assessment of risks to soil invertebrates and plants and CCME guidelines were used to assess toxicity of fresh crude to livestock (Stephenson et al. 2015).

As discussed in Chapter 4, graphs of predicted toxicity using the TLM versus observed chronic toxicity indicate very large variance s (10-fold or greater). The versions of the TLM used by Stephenson et al. (2015) have been super seded by newer publications; however the newer TLM predictions are still imprecise and are based on many untested assumptions.

Predicted total PAH concentrations in sediment pore water were less than 0.2 µg/L after four weeks and less than 0.05 µg/L after 1-2 years. Predicted river water TPH concentrations were mostly less than 0.01 mg/L. No significant risks were predicted for fish or benthic invertebrates, assuming additive toxicity of the various hydrocarbon compounds (Stephenson et al. 2015). The except ion was in the Chickadee Creek scenario where fine -grained sediments were predicted to create conditions leading to higher concentrations in sediment pore water. The authors stated that in gravels most likely to be used by salmonid fish as spawning habitat s, the expected average TPAH concentrations were below benchmarks established to identify potential for blue sac disease or mortality of developing eggs (Chapter 4).

Although the statement regarding spawning gravels is logical if oil droplets are not entra ined in sediments by hyporheic flows, it does not acknowledge that some fish species (including salmonids such as mountain whitefish) may use finer-grained sediments for spawning. Observed toxicity to lake whitefish

and northern pike embryos from exposure to PAHs in near-shore sediments of Wabamun Lake four to seven months after the spill of heavy fuel oil illustrates this point.

Stephenson et al. (2015) discussed the possibility that crude oil could become entrained into and trapped in river gravels as a result of hyporheic flows (Figure 2.3). They speculated that although this fate is possible it would depend upon:

- Exchange between river water and subsurface pore water. This can occur within river reaches at scales of hundreds of metres to kilometres, or locally at the scale of a single salmon redd. Depending upon the season and local topography, rivers may alternate locally and seasonally between 'gaining' and 'losing' water from or to groundwater of hyporheic system;
- Whether upwelling dominates - for rivers like the Kitimat it is likely that hyporheic flows will be dominated by inflowing groundwater and that such groundwater will be upwelling through the gravels, thus limiting the potential for ingress and retention of crude oil; and
- Whether crude oil becomes dispersed by turbulent river flow as fine droplets that would not rapidly resurface and would be available to become entrained into hyporheic flow should conditions allow—such dispersion could occur for condensate although it has low density and is likely to rapidly resurface and could occur for synthetic oil, but is less likely for dilbit, which is more viscous. Stephenson et al. (2015) noted that the magnitude of predicted oil spill effects at the population level would depend upon the timing and location of the spill with respect to fish and their key habitat, as well as the availability of unaffected habitats either upstream or in tributaries.

The Panel notes that fish demonstrate habitat fidelity so even if there were unaffected habitats, the fish might not use those habitats but rather remain in the contaminated areas. This was the concern in the Pine River spill (Birtwell 2003); however, follow-up monitoring of population-level indicators in the Pine River was insufficient to either confirm or reject this hypothesis.

The Panel understands the emphasis on sensitive salmonid species when conducting risk assessments since the standard practice is to select the lowest effects thresholds for the contaminants of concern (and these are usually for salmonids). However, other species, such as white sturgeon or eulachon, would be more vulnerable if the last few remaining spawning habitats were oiled or the food supply was reduced in critical rearing habitats. Furthermore, species at risk would be less resilient to the loss of a year class (or even a portion of a year class).

8.17 Assessment of Risks of Hypothetical Spills from the Proposed Trans Mountain Pipeline Expansion Project

8.17.1 Semi-quantitative assessment of spills at selected locations

Kinder Morgan (2013) conducted a semi-quantitative risk assessment of the Trans Mountain Pipeline Expansion Project (TMPEP). Quantitative estimates of failure frequency and qualitative estimates of consequences were combined and applied to hypothetical spills at selected locations along the pipeline route. Risk results were used to identify the need for additional pipeline design measures to reduce the probability and/or the consequences of failure (e.g. relocation of a valve site to reduce potential outflow in a high consequence area). A reliability approach was used to estimate failure rather than incident statistics. Areas with higher potential consequences were identified based on land use or location with respect to water bodies.

A worst case full-bore rupture was used for the TMPEP assessment; however, as for the NGP assessment, the elapsed time to achieve valve shut-down was short—in this case only 10 minutes. The TMPEP

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authors stated that 10 minutes for a readily identifiable catastrophic rupture is conservative because the Control Centre Operation would recognize the event immediately.

The Panel has the same comment as for the NGP a ssessment—the assumption of 10 minutes to valve shutdown significantly (and possibly unrealistically) constrains the spill volume, thus fundamentally affecting the results of the risk assessment.

Recommendation: The Panel suggests that risk assessment scen arios that assume a more likely combination of factors (less than full-bore rupture with a longer elapsed time to response) would be more relevant and comparable to actual spill cases. These scenarios would also be viewed as more credible by stakeholders sensitized by recent spills that experienced long delays before shutdown.

Eight case studies were assessed in the TMPEP report to provide relevant information for use in developing assumptions and inputs to the qualitative assessment of consequences. The cases included the Kalamazoo River, Wabamun Lake and Pine River spills. All of the case studies used are presented in Table 8.2.

Table 8.2 Case Studies Considered for the Trans Mountain Pipeline Expansion Project (TMPEP) Assessment of Environmental Effects of Oil Spills

Oil Spill	Location	Year	Spill Source	Oil Type	Volume (m^3)
Kalamazoo River	Michigan, US	2010	Pipeline Full- bore Rupture	Diluted Bitumen	3,200
Wabamun Lake	Alberta, Canada	2005	Rail Accident	Bunker "C"	712
East Walker River	California/Nevada, US	2000	Truck Accident	Bunker "C"	14
Pine River	BC, Canada	2000	Pipeline Full- bore Rupture	Light Crude	985
Yellowstone River	Montana, US	2011	Pipeline Full- bore Rupture	Light Crude	240
OSSA II	Bolivia, South America	2000	Pipeline Full- bore Rupture	Mixed Crude	4,611
DM932	Louisiana, US	2008	Barge Accident	Bunker "C"	1,070
Westridge	Burnaby, BC	2007	Pipeline Third Party Damage	Heavy Synthetic Crude	224

The spill scenarios involved releases into the Upper Athabasca, North Thompson and Lower Fraser Rivers. "Credible worst case spill volumes" ranged from 1,250 to 2,700 m^3. The crude spilled was Cold Lake Winter Blend dilbit (see Table 2.2).

Recommendation: Differences and similarities between the assessed consequences of a dilbit spill into various locations along the Athabasca, North Thompson and Lower Fraser Rive rs and case study results should be used to develop future research projects as well as guidance for standardized monitoring of oil spills. Some suggested topics for further investigation are indicated in italics.

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The qualitative assessment of consequences varied with season and spill location. For locations on the Athabasca River and Thompson River, consequences in winter were assessed as low (except for localized effects on aquatic invertebrates) because many of the receptor organisms would be dormant and overland flow would be slowed with some oil absorbed into the snowpack. Furthermore, it was assumed that environmental effects may be minimized because most of the oil is recoverable. The consequences in summer in the Athabasca and Thompson Rivers were predicted to be much greater because of rapid dispersion due to high flows, entrainment due to turbulence, and deposition along shorelines. In addition, the TMPEP assessment assumed that as the oil is transported downstream and weathers, it becomes more viscous and dense and interacts with shoreline sediments, e.g., forming OPAs and resulting in oil that submerges in low energy areas such as eddies and backwaters. The TMPEP authors stated that although water in the Athabasca and Thompson Rivers is somewhat turbid, the suspended sediment load is not particularly high, little oil would be entrained in the water column, and the water has no appreciable salinity. Thus, OPA formation would not be a dominant factor in the fate of the spilled oil. The extent of dispersion was assumed to be greatest in the summer, extending as far as 100 km downstream. The magnitude of effects on aquatic biota ranged from low to high depending upon the group and the proximity to the spill.

Consequences in spring or fall were considered to be intermediate in the Athabasca and Thompson Rivers due to lower and less turbulent flows than during the summer. Some submerged oil was assumed to be present in eddies and backwaters, as was predicted for summer conditions. The extent of downstream dispersion was assumed to be about 25 km.

Actual seasonal differences in the consequences of spills may differ from those predicted by the TMPEP assessment. For example, winter time spills involve much slower loss of volatiles and a longer time for dispersion to occur, and recoverability of a winter spill will depend upon the logistic challenges associated with the spill location.

The conclusion of the TMPEP authors that OPA formation would not be a dominant factor in the fate of the spilled dilbit is subject to debate. If a spill occurs during high flow when suspended sediment is elevated and/or if the flow-path involves interaction with riparian and floodplain materials (as was observed in the Kalamazoo spill), OPA formation could be much more significant.

The Fraser River has ice-free conditions during the winter; thus, predicted winter consequences in the Fraser River differed from those predicted for the Athabasca and Thompson Rivers. While some receptors would be dormant, birds such as bald eagles and some waterfowl could be present and potentially experience high magnitude consequences. High to medium magnitude consequences were predicted for aquatic receptors in side channels.

Summer consequences in the Fraser River location also differed because a side channel at this location would be heavily affected during high flows. Much of the oil was predicted to become stranded along shorelines and in riparian areas. The potential for OPA formation and submerged oil in the side channel may have been underestimated given the experience with the Pine River, Kalamazoo and Wabamun spills.

Spring and fall consequences were also predicted by the TMPEP assessment to be greater in the Fraser River location because flow conditions would disperse the oil as much as 60 km downstream during a time when many migratory birds would be present and semi-aquatic mammals would also be present and active. Effects were assumed to be insufficient to affect regional bird populations. *However, this conclusion may not apply to species-at-risk, to situations where a population has insufficient resiliency to withstand the loss of a substantial portion of breeding pairs, or if sublethal effects of residual oil on food supplies essential to migrating birds results in reduced fitness.*

In summary, the qualitative assessment of consequences of modeled dilbit spills in the TMPEP report was reasonably consistent with case study data, with appropriate conservatism added to account for uncertainty. The potential for OPA formation may have bee n underestimated given the experience with the dilbit spill into the Kalamazoo River, as well as the behaviour of heavy fuel oil in the Wabamun Lake spill. The recovery period was assumed to be from 12 months to five years post-spill for almost all of the spill scenarios. No evidence was presented in support of this assumption; however, the limited information on recovery from case studies suggests that this range is reasonable for most receptors; e.g. fish in the Pine River (Section 8.8).

8.17.1.1 Stochastic modeling of spills at locations near and at the mouth of the Fraser River

<u>Recommendation:</u> The results of the stochastic oil spill fate and transport modeling for a spill at the Fraser River and Delta location should be used as input for spill preparedness p lanning, as well as for identification of priority requirements for verification of the model.

The TMPEP assessment included stochastic oil spill fate and transport modeling in support of the risk assessment for a spill at the Fraser River and Delta locati on near the Port Mann Bridge. Stochastic modeling was performed due to the complexity of the situation at that particular hypothetical spill location. The modeling results by season provide valuable insights regarding the most probable consequences and imp lications for cleanup. The results are summarized below, accompanied by comments from the Panel, which are in *italics*.

Winter: >80% of oil is stranded within three days of spill; 11% evaporates; < 5% remains on the surface; < 1% is submerged, biodegrades or dissolves; < 0.1% forms OPAs.

- *The implications are that the spill response will be focused on stranded oil, with potential for substantial disturbance of habitat during the cleanup. Also, if most of the oil cannot be remove d before higher spring flows, re-mobilization might occur.*
- *The assessment assumed that many of the receptors are absent or dormant in winter —this would be true for plants (dormant) and some (but not all) invertebrates, and perhaps for amphibians, but many birds overwinter at this location.*

Summer: < 60% of oil is stranded, about 10% evaporates; about 30% remains on the water surface; < 1% is submerged, biodegraded or dissolves; < 0.1% forms OPAs.

- *There is still a substantial amount of stranded oil, resulting in medium to high effects on riparian vegetation and in marsh areas; trade-offs between the effects of the stranded oil and the effects of the cleanup will require very careful evaluation.*
- *The oil remaining on the water surface is predicted to result in medium to high effects on ducks and geese and high effects on semi -aquatic mammals; which illustrates the importance of rapid and effective removal of floating oil.*
- *OPA formation may have been underestimated, given the sediment load of the Fraser River.*

Spring and Fall: 70% of oil is stranded; 10% evaporates; 20% remains on the surface; a small amount of oil (< 1%) is predicted to become submerged, undergo biodegradation or dissolve; OPA formation is not predicted (< 0.1% of oil volume).

- *There is an implied assumption that spring and fall represent one condition, which does not account for the highly stochastic weather, discharge, ice state and biological events that typify these seasons. One condition does not encompass the range of conditions, nor the chan ge in conditions during or immediately after a spill. <u>Thus, a source of significant uncertainty does not</u>*

seem to be recognized. The same argument applies to winter and summer, although the size and frequency of changes are not as large.

- *The assessment once again focused on the effects of recovery of stranded oil. This common theme again highlights the importance of having a sufficient understanding of the receiving environment to enable confident evaluation of the trade-offs between the effects of residual oil and the effects of further cleanup.*
- *The TMPEP assessment assumed that riparian habitat along the mainstem of the Fraser River would be remediated with less intrusive methods and a greater emphasis would be placed on natural attenuation of low levels of residual oil. This is a debatable assumption given the potential effects of the modeled spill on socioeconomic and cultural values. The Panel suggests that the net environmental benefit calculations would be challenging and subject to considerable uncertainty.*
- *High effects on semi-aquatic mammals (due to oiling) were predicted and medium to high effects on ducks and geese were predicted (including reproductive effects caused by transfer of oil to eggs or effects on habitat quality caused by disturbance arising from oil spill response efforts).*
- *Particular attention was paid to the western sandpiper. Hundreds of thousands of these birds may congregate in the assessment area, feeding on biofilm and invertebrates —the fate modeling showed that the probability of oiling areas used by the birds is very low —therefore the sandpipers are unlikely to be significantly affected and any effects are likely reversible.*
- *The Panel notes that the reliability of the TMPEP fate model is paramount regarding risk to western sandpiper.*

Recommendation: The Panel recommends the use of stochastic fate modeling for complex, sensitive receiving environments because of the additional important information provided to decision-makers regarding the probability of the various levels of effect on specific VECs. This information can assist decision-makers in focusing preparedness and response plans. Stochastic modeling also provides a basis for prioritization of environmental data collection.

8.17.1.2 Comparative assessments of Unmitigated and Mitigated Spills at the Westridge Marine Terminal

Assessment of an unmitigated and a mitigated spill of heavy synthetic crude oil blend at the Westridge Marine Terminal produced a mass balance comparison, presented in **Table 8.3** (Kinder Morgan 2013).

Table 8.3 Mass Balance Comparison for a 160 m³ Spill at the Westridge Marine Terminal

Amount (m³)	Unmitigated Case	Mitigated Case
On shore after 1 day	16.6%	10.7%
Left on water after 1 day	0.4%	0.1%
Evaporated after 1 day	1.2%	0.9%
Dissolved after 1 day	1.8%	1.2%
Biodegraded after 1 day	< 0.1%	< 0.1%
Inside the containment area but not yet recovered	80%	0%
Recovered inside the containment boom	n/a	80%
Recovered at sea	n/a	7.1%

After one day, no oil was predicted to remain on the water with mitigation, compared to 80% if unmitigated (although oil inside the containment area would be recovered over subsequent days).

Mitigation reduced shoreline oiling but did not eliminate it (10.7% on the shoreline with mitigation versus 16.6% on the shoreline without mitigation).

Recommendation: The Panel urges incorporation of lessons learned from incidents as well as assessments conducted for proposed project into preparedness and response planning by responsible authorities and industry. The Panel notes that common themes among incidents and assessments include communication, coordination, adequate resourcing and training.

The authors noted that the evaluation of responses to a spill at the Westridge Marine Terminal contributed to the following 'lessons learned' (these lessons learned are in addition to those presented in Section 8.11).

- Proximity of ready response capability to a spill site together with site-specific response plans (which responders have exercised) help to greatly increase the effectiveness of response. In the case of the simulation study, the modeling runs helped WCMRC gain a good understanding of key requirements to effectively improve response. This is similar to the iterative learning achieved through oil spill exercises.
- There is no substitute for establishing an early line of defense by rapidly booming a release or damaged vessel, when possible. This knowledge is tempered by the reality that health and safety conditions, suitable nearby anchoring sites and operational constraints may not always make this outcome possible.
- Recovery assets should be located in relatively close proximity to the spill, as would be the case for Westridge Marine Terminal.
- Use of model-derived trajectory and slick thickness information to direct skimmers can help identify optimum recovery locations. While remote sensing offers considerable opportunities for spill detection, it also has limitations. The combination of numerical modeling and remote-sensing data provides the most powerful approach to both current and future predictions of slick positions.

The WCMRC has proposed a harbour base that would be continuously staffed if the TMPEP is approved (Kinder Morgan 2014).

The Panel notes that some of the above lessons learned would also apply to the *Marathassa* spill which occurred in English Bay in 2015 (Section 8.12).

8.18 Assessment of Accidents and Malfunctions for the Energy East Pipeline Project

The Panel reviewed the assessment of accidents and malfunctions for the proposed Energy East Pipeline project (EEPP), which included consideration of the fate, transport and effects of three representative crude oils: Bakken crude oil, Husky Synthetic and Western Canadian Select diluted bitumen (Table 2.2). Four categories of rivers and streams were assessed based on magnitude of flow and width.

EEPP assumptions included:

- The release of the entire volume of the spill directly into the water body;
- Complete, instantaneous mixing; and
- The entire volume of Constituents of Interest dissolved into the water column (Energy East 2014a).

These assumptions were referred to as 'conservative'. However, the spill case studies illustrated that release to a terrestrial flow path leading to a watercourse can have significant consequences in terms of fate and behaviour of spilled oil as well as consequences to receptors along the flow path (see also the

TMPEP assessment). Complete, instantaneous mixing and complete solubilisation are not conservative assumptions because there would be no floating oil to cause acute mortalit y of birds and semi -aquatic mammals. These assumptions do not consider acute effects of floating oils or other important fate mechanisms that have been shown to significantly affect consequences (e.g., the formation of sinking oil).

The EEPP assessment was based on simplistic estimates of hydrocarbon concentrations, reliance on low likelihood of a spill occurring at particular locations, and blanket assurances regarding mitigation, which is illustrated by the following quote:

> "The environmental effect of a crude oil spill would vary both temporally and spatially depending on the volume, timing and location of the spill. Localized effects could occur from virtually any size of crude oil spill. In general, the chance of a spill occurring at any discrete locati on along the pipeline or marine transportation of oil is low and if a spill were to occur, it likely would be relatively small (4 bbl or less). In the event of a spill, Energy East would respond according to applicable regulations and their Environmental R esponse Plan which will contain provisions for protecting and mitigating potential effects to environmental receptors. In addition, in the event of a spill, Energy East will consult with regulatory agencies to determine the appropriate and preferred approach to cleanup and monitoring" (Energy East 2014b).

The Panel was initially encouraged to see that the EEPP assessment considered locations along the pipeline route that are rarely evaluated in the context of oil spills; e.g. the South Saskatchewan River, t he Red River, a lake in Ontario (Trout Lake), the Rideau River, and the Iroquois River, NB (Energy East 2014b). These locations represent a wide variety of receiving environments with important differences that can have substantial effects on oil fate, behaviour and toxicity. Unfortunately, the simplistic nature of the assessment did not address these characteristics in any meaningful way (e.g. , sediment loads in the South Saskatchewan River, morphometry of Trout Lake). Instead, the same statement that is q uoted above was repeated for each location, with slight (if any) variations.

Statements regarding the low probability of a spill were also very similar for all assessed locations. The rationale for low probability included such points as:

- The Project's pip eline materials and design, as well as the strategic placement of valves, are expected to minimize the likelihood of a spill and reduce the volume of oil released in the event of a spill;
- Rivers will be crossed by horizontal directional drilling, which wil l reduce the probability of a rupture by reducing threats to the pipeline given the depth of the pipe below the river. The depth and overlying materials also would help confine a release and reduce the possibility of the crude oil reaching the river;
- Valves are strategically located along the EEPP route to reduce the amount of crude oil that potentially could be spilled. The location of valves, spill containment measures and emergency response procedures would mitigate adverse effects to both surface water and groundwater; and
- The assessment used conservative assumptions to overestimate risk. Many assumptions, such as all benzene from the oil would be instantly dissolved into the water, are unrealistic but help to screen for the potential for effects.

The Pa nel found the EEPP assessment to be overly simplistic, with blanket assurances regarding mitigation, generalized assumptions that mitigation would always be effective, and a reliance on low probability to reduce risk. The claim that the assessment used con servative assumptions to overestimate risk is incorrect since assuming instantaneous mixing of a spill would be conservative with respect to fish but liberal with respect to birds and semi -aquatic mammals. The EEPP assessment did not add any value with respect to risk-based prioritization of mitigation measures along the pipeline route.

<u>Recommendation:</u> The Panel suggests that the sites of interest used in the EEPP assessment merit being re-analysed using analytical tools similar to those used for Trans Mountain or Northern Gateway pipelines—particularly with respect to fate and transport and the level of sophistication with respect to acute and chronic toxicity, shoreline effects and overland flow effects with season.

8.19 Summary Comments and Recommendations Regarding Risk Assessments of Oil Spills within Environmental Impact Assessments

The NGP and TMPEP risk assessments are impressive documents in their scope and detail and are among the best of their kind. The fate modeling in support of the NGP assessment and the use of stochastic fate modeling for particularly sensitive and complex receiving environments done for the TMPEP assessment provided valuable details with respect to the relative proportion and location of floating, dissolved, entrained, stranded and submerged oil. The use of case study information to develop a qualitative effects assessment for the TMPEP provided a useful alternative to the calculation of quantitative hazard indices using single point-estimates of effects thresholds.

Although the NGP and TMPEP risk assessments had several strengths, they were subject to the same uncertainties and challenges as any large-scale assessment. The results of the assessments can be used as valuable input in identifying priorities for preparedness and responses as well as for focusing on critical uncertainties associated with factors which drive the risk. However, the results of the assessments cannot be assumed to be reliable predictors of what will actually happen during a spill. The state-of-the-science of risk assessment has not yet achieved the required level of verification via comparison of predicted versus observed effects. Furthermore, uncertainty in risk assessment will always be substantial, and in some cases irreducible. "Personal judgment and expert opinion will necessarily be a part of such risk assessments because they handle rare events in complex systems" (Hague et al. 2014).

The development of the scenarios to be used is one of the most difficult challenges in risk assessment, and the NGP and TMPEP assessments illustrate this challenge. In both cases, a full bore pipeline rupture was assumed and described as "worst case". However, the legitimacy of the description of the risk scenarios as "worst case" is subject to question because of the assumed short time period between detection of a pipeline breach and shutting off of valves to stop the flow of oil. Other sources of uncertainty regarding the "worst case" nature of the risk scenarios include:

- Lack of assessment of winter spills for the NGP;
- Choices of spill sites which, in some cases, reduced the amount of oil reaching the most sensitive areas (e.g. reducing oil reaching the estuary at Kitimat) (Hodson and Martin 2012);
- Inclusion of receptor species that may be sensitive because of their habitat, conservation status or cultural value (e.g. eulachon) (Hodson and Martin 2012);
- The use of average physical and chemical characteristics of river receiving environments (Hodson and Martin 2012); and
- Missing or inadequate representation of potentially important processes, such as hyporheic flows.

The development of risk scenarios is a fundamental issue because they represent high stakes, uncertain facts and conflicting values (Hauge et al. 2014). As Hauge et al. point out, uncertainty makes different interpretations possible, and values may be embedded in knowledge production and interpretation (as discussed in Section 8.6 with respect to the EVOS). The 'appropriate' size, location, flow-path, time-scale, receiving environment and receptor species for each oil spill assessment is subject to debate, not only regarding which particular combination might best represent a worst case, but also whether a worst case is what is needed to support decisions.

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In some contexts, worst case scenarios are so highly unli kely as to be of little value for decision -making. It may be more useful to choose more likely scenarios which occur more frequently.

Reduction of some of the uncertainties associated with risk assessment is possible. For example, including migration patte rns and other behavioural characteristics (e.g. fidelity to feeding grounds and spawning sites) and combining these patterns with stochastic dispersion modeling would offer new insights into the potential exposure of aquatic receptors. Unfortunately, data required for more sophisticated spatially and temporally -explicit exposure modeling is often lacking. The use of chronic toxicity data from the most relevant test species, combined with field information from spill case studies would produce less uncertainty than using acute:chronic ratios. Careful attention to all exposure routes for each receptor helps ensure that exposure is not underestimated.

Several sources of uncertainty are very difficult to reduce. Major oil spills are rare and each of these rare events is unique - making generalizations from one case to another very difficult. Stochastic processes such as weather influence the fate and effects of oil spills in unpredictable ways.

The framework and tools for ERAs originally were developed for relatively small sites; their applicability to 1,000's of kilometres of pipeline crossing highly variable habitats with major physical and chemical gradients is questionable. Of necessity, risk assessors must select representative sites along these routes and th e assessed risk is ultimately a compari son of model ed concentrations of hydrocarbons in water, sediment (or sometimes tissue) to a single benchmark of toxicity. Rarely, modeled concentrations are compared to a dose -response curve. Effects of exposure to se veral chemicals are usually assumed to be additive, although the validity of this assumption is highly questionable, particularly when chemicals with very different modes of action are combined. In particular, the interactive effects of oil and municipal and industrial effluents have not been considered for risk assessments in freshwater and coastal ecosystems. The existing condition of the receptor is almost never explicitly considered; e.g. , if a fish population is already exhibiting poor condition due to the existence of other stressors in the receiving environment.

Given the limitations of the toxicological approach to risk assessment, there have been many calls for the incorporation of ecology into risk assessment. The use of s patially-explicit exposur e mode ling that incorporates seasonal habitat use by each receptor and superimposes contaminant gradients or patterns has been a significant advance; however, such modeling rarely appears within impact assessments. More commonly, a weight-of-evidence approach is used, which combines laboratory, field and modeling lines of evidence into an overall risk estimate. The evaluation of effects of exposure to multiple stressors is the subject of active research; however, the results of this research seldom arereflected in risk assessments.

In summary, although many improvements are needed in both exposure and effects evaluations within risk assessment, the most fundamental challenges centre on the framing of the risk assessment. Risk framing can be from the perspec tive of informing management, or it can be from the perspective of deciding whether the risk is acceptable to society (Hague et al. 2014). Public concerns about impacts may be very different from a petroleum company's concerns. Often, risk assessments are part of processes where questions can be asked of science and yet cannot be answered by science (Weinber g 1972 cited in Hague et al. 2014). Risk assessments cannot provide the level of confidence often demanded by the public and political decision -makers—to achieve the required confidence would take decades of research, and even then, we may not be able to adequately represent the complexity of ecosystems.

8.20 Assessment of Risks Associated with Transport of Crude by Rail

A major increase in crude-by-rail transport started around 2009. Since then, crude-by-rail transport in the US has increased by 40-fold or more (Etkin 2015). The bulk of the oil is crude oil from the North Dakota Bakken fields and the Alberta oil sands. A 42% increase has occurred in average an nual spillage of oil by rail in the US but at a reduced rate per volume transported (Etkin 2015) (Table 1.4). However, accidents

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involving crude-by-rail are of particular concern because of the potential for fires and explosions and impacts on urban areas traversed by rail lines.

Etkin (2015) cited analyses showing a steep decline in the frequency of derailments per train mile, with about one mainline derailment for every 3.85 billion tons of freight (including but not limited to crude-by-rail). It has been estimated that about six crude-by-rail related derailments would occur annually. Consequences were determined by oil type, volume, location and timing.

For Bakken crude, the property of greatest concern is volatility leading to explosion and fire, e.g. the Lac Mégantic, QC, disaster. For diluted bitumen the greatest public concern is that oil will become submerged when it reaches a body of water. Bitumen blends vary considerably depending on the source, blending procedures and diluent used (Table 2.3); therefore, the potential for sinking will vary. However, the Kalamazoo spill and the results of modeling in support of risk assessments illustrate that sinking is a definite possibility when dilbit is spilled, depending upon weathering-related density changes, turbulence, salinity and the amount of sediment in the water.

Estimated spill volumes for crude-by-rail Unit Trains are presented below (from Etkin 2015). The ranges are based on a less conservative 9.2% release rate to a more conservative 16.7% release rate:

- 25th percentile – 21 to 38 m^3 involving 2 cars
- 50th percentile – 52 to 95 m^3 involving 5 cars
- 75th percentile – 104 to 190 m^3 involving 10 cars
- 90th percentile – 178 to 322 m^3 involving 17 cars
- 95th percentile – 240 to 436 m^3 involving 23 cars
- Worst case – 1,274 to 2,313 m^3 involving 122 cars

In actual crude-by-rail spill cases in Canada and the US in 2013 and 2014, the lowest volume spilled was 17 m^3 and highest was 2,909 m^3, which was greater than the estimated worst case (Etkin 2015).

The higher percentile spill volumes (and the highest actual spill volume) are substantial spills, comparable to tanker spills and the larger pipeline spills. Therefore, preparedness and response plans should not discount the potential for higher-volume spills from crude-by-rail accidents. Challenges related to response to spills of Bakken crude transported within Canada include:

- Fires in populated areas or in locations where wildfires may be triggered; and
- The adequacy of first responder systems in locations that have seen a sudden increase in crude-by-rail (e.g., from the Estevan area of south-eastern Saskatchewan).

The Canadian Association of Fire Chiefs is reviewing the preparedness for crude-by-rail spills (Etkin 2015).

Responses to Bakken spills have included the use of sorbent and containment booms along with water spraying to corral oil for skimming and vacuum pumping. Sorbent pompoms and pads have been used in some areas (Etkin 2015).

Response to spills of dilbit would include the requirements for general preparedness for inland locations, consideration of the flammability of diluent, and development of capabilities for submerged oils.

Etkin (2015) noted that risk mitigation through prevention is key. Prevention measures would include training of railway personnel and first responders, positive train control, improvements in braking systems, and track inspections and maintenance. Newer tank car designs would reduce the potential for leakage or spillage. Reducing volatility of the oil (e.g., via conditioning of Bakken crude) would be another preventive measure. Canada and the US recently jointly announced forthcoming wide-ranging

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changes to rail safety regulations including braking systems, speed limits, improved tank car specifications, and rules regarding securing of unattended trains[8].

8.21 Overall Conclusions Regarding Assessment of Risks of Oil Spills

Assessments of the risks of hypothetical spills conducted in support of the environmental impact assessments of proposed pipeline projects in Canada have used unrealistic assumptions regarding spill response time, based upon experience to date. The more detailed and comprehensive risk assessments, such as those conducted for the proposed Northern Gateway Pipeline, were sufficient for risk-based prioritization of preparedness and response planning as well as mitigation via pipeline design. However, the consequences were likely underestimated given the assumption that it would take only a few minutes to recognize a pipeline breach and deploy shutoff valves.

The review of risk assessments also revealed a wide "range of practice" applied to assessments by different practitioners. The Panel was concerned about the breadth of this range, particularly since the range was demonstrated in submissions to the same regulator (the NEB). Therefore, in addition to research and monitoring to provide information needed to support risk assessments, there appears to be a need for an explicit 'standard of practice' regarding the assessment of risk of oil spills. This is particularly important given the high level of concern associated with the production and transport of oil.

There is an urgent need for investigations in Arctic Canada regarding the fate and behaviour of oil under Arctic conditions, the sensitivity (and resilience) of Arctic environments and the effectiveness of available spill response measures. Arctic marine systems include unique habitats, including the epontic habitat. Epontic communities are currently the subject of research; however, the potential effects of oil spills on these important communities should be investigated. The results of Arctic marine research should be consolidated periodically to serve as a baseline for monitoring activities (SL Ross 2014).

Investigations of oil spill effects must include collaboration with Indigenous peoples to ensure that traditional knowledge is incorporated into our overall understanding of the risks of oil spills.

There is a critical need for a coordinated and integrated database of information relevant to the assessment of risk of oil spills in Canada. The data needed for input to risk assessments in Canada are often either absent or widely scattered among government agency, industry, and academic sources. The highest priority should be accorded to assembling, and if necessary, sampling to build an understanding of pre-spill conditions at locations that are under the highest threat of contamination. The Panel suggests that efforts initially focus on geographical areas at risk that have received less attention with respect to oil spills (e.g. inland rivers, portions of the Great Lakes with the highest oil transport-related activities, and the Gulf of St. Lawrence).

There is a compelling need for coordinated environmental sensitivity assessment in Canada using standard tools and analytical approaches. This assessment must include integration of traditional knowledge and information about traditional uses of resources. The gap in sensitivity mapping is larger in inland Canada. Some industries are conducting their own sensitivity mapping; however, this information is generally proprietary. Sensitivity mapping efforts should be prioritized using screening tools that can identify the most likely sensitive watercourses subject to oil spills because of their location in proximity to production or transportation routes, characteristics that would affect fate and behaviour in such a way that risks are increased, and the presence of sensitive environmental components. Partnerships among industry, government and academia will be necessary if adequate coverage of receiving environments in Canada is to be achieved.

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[8] *e.g., http://www.cbc.ca/news/canada/montreal/lisa-raitt-u-s-counterpart-announce-broad-new-standards-for-rail-safety-1.3057150 (May 1, 2015).*

Regarding the potential for spills in the Canadian Arctic, SL Ross (2014) suggested:

- Consolidate the rapidly expanding knowledge about the aspects of the ecology of Arctic species relevant to monitoring (spatial distribution, populations, habitat use, seasonality);
- Prepare a review of research into effects of oil spills on Arctic marine and anadromous fish species (e.g., Arctic and polar cod, Arctic herring, Arctic char, Arctic cisco). *The Panel adds the need for consultation with Aboriginal peoples regarding their knowledge of fish species and the areas most sensitive to oil spills, e.g., char spawning areas;*
- Develop a decision tree to help decide whether a field monitoring program is required during a spill incident. *The Panel suggests that field monitoring would be required for all but small spills; therefore, the decision tree could be designed to assist in the determination of the scope of the monitoring program; and*
- Prepare a readily accessible up-to-date directory for all regional environmental emergency coordination centres in Canada, including the names and contact information for the senior personnel. *The Panel strongly endorses this recommendation.*

In addition to the above more general recommendations, SL Ross (2014) suggested that it would be useful to develop an understanding of the sensitivity of Canadian species to PAH-induced immunosuppression and the potential long-term consequences of it in Canadian fish populations during spills.

Focussing research and management on ecological resilience would be useful both in principle and for specific spill scenarios. As the NRC (2013) pointed out, resilience provides a valuable conceptual framework for managing complex systems because it focuses attention on how systems are affected by short-term disturbances and long-term stresses. Resilience can be either increased or decreased through management decisions. Analysis of the trade-offs inherent in management decisions would benefit from the use of a resilience framework.

Adopting resilience as a fundamental framework will require much more attention to the acquisition of baseline data, both for the natural environment and for the socioeconomic environment

The Panel suggests that our focus should be on prevention of large spills and rapid and effective response to smaller ones because we will never be able to eliminate spills. Therefore, the Panel **recommends:**

- Identifying where most of the spills are occurring and why (e.g., pipeline spills into wetlands are more common than into rivers; truck spills are more likely to enter storm sewers and then rivers rather directly impact rivers; rail lines often run parallel to rivers and derailments may occur more frequently with certain load and geographical configurations, etc.);
- Examining past spill response records, plus the current risk management processes and regulations and their effectiveness and then honing in on the weak spots;
- Filling in critical data gaps regarding environmental sensitivity;
- Conducting relative risk assessments of the common spill location types overlain on the most sensitive receiving environments; and then
- Identifying the combination of spill source, receiving environment and level of preparedness that most urgently requires attention.

8.22 Research Needs and Recommendations

The following list presents knowledge gaps and corresponding recommendations, in order of priority.

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8.22.1 High priority research needs (short-, medium-, and long-term)

8.22.1.1 Gap: Baseline information

<u>Recommendation:</u> Collect and Evaluate Baseline Information from High -Risk Areas (Short-term and Extending to Long-Term)

Research is needed regarding the current status of sensitive species and vulnerable habitats for specific, pre-defined locations in Canada representing a range of human disturbance, from relatively undisturbed to highly disturbed. This information can then be used as the base case for assessment of any future spill. The priority locations for collection of baseline information are those identified as high -risk in recent assessments conducted for Transport Canada (WSB and SL Ross 2014a,b), combined with information on areas with proposed new human activit ies that may increase the likelihood of spills and/or contribute to cumulative effects of spills (e.g. , LNG terminals along the British Columbia coast). Input from Indigenous peoples and from key stakeholders should be part of the selection process for bas eline collection areas.

8.22.1.2 Gap: National, accessible databases for use in assessment of oil spills

<u>Recommendation:</u> Establish a National Database of Information Relevant to Risk Assessment of Oil Spills, Preparedness and Response (Short-Term)

<u>Baseline Data:</u> There is a critical need for a coordinated and integrated database of information relevant to the assessment of risk of oil spills in Canada. (e.g., data collected for EIAs, EEM programs, compliance monitoring (e.g., for municipal or industrial discharges), research programs, etc.). The Panel is aware that some databases are already established (e.g. , for EEM data); however, there is no readily accessible, national portal to metadata on freshwater systems. Since the assembly of such a database would be a huge undertaking, the Panel suggests that efforts initially focus on geographical areas at risk that have received less attention with respect to oil spills (e.g., inland rivers, the Great Lakes).

The Panel strongly recommends that the national database include data for the Arctic.

<u>Experimental and Monitoring Data:</u> To support predictive numerical models and operational guidelines for spill response , there is a need for a rigorous database on the fate, behaviour and effects of various types of oil spilled and the efficacy of current and emerging oil spill countermeasures over a range of environmental conditions.

8.22.1.3 Gap: Shoreline sensitivity in the Arctic

<u>Recommendation:</u> Extend Shoreline Sensitivity Mapping to Selected Arctic Locations (Short-term to Medium-Term)

The Panel strongly supports the continued mapping of shoreline sensitivity and calls for this mapping to be extended into Arctic areas.

Shoreline mapping should be conducted in areas where oil and gas exploration and development is already occurring or is expected, e.g., Beaufort Sea and the Mackenzie Delta. The mapping should also consider other human activities that would affect relative risk, such as mining, where large port facilities have been proposed involving not only transport of the minerals out but fuel in, and increased commercial shipping traffic through the Northwest Passage, which brings greater risk of fuel oil spills.

8.22.1.4 Gap: Shoreline sensitivity in inland Canada

Recommendation: Extend Sensitivity Mapping to Inland Freshwater Habitats (Short-Term to Medium-Term)

The current lack of inland shoreline sensitivity information in the open literature is a serious gap affecting the ability to apply risk-based preparedness and response planning. Prioritization of areas requiring mapping could be based upon evaluation of the intensity of human use, the current knowledge of the relative sensitivity of ecosystems and the availability of information from various sources. For example, high-priority areas for mapping might include areas upstream and downstream of hydroelectric dams.

Sensitivity mapping should be coordinated and should use standard tools and analytical approaches. This assessment must include integration of traditional knowledge and information about traditional uses of resources.

8.22.1.5 Gap: Efficacy of oil spill responses for various types of oil

Recommendation: Conduct Experiments on Efficacy of Current and Emerging Oil Spill Countermeasures (Short-Term to Medium-Term)

More data are required regarding the efficacy of current and emerging oil spill countermeasures for various types of oil spilled over a range of environmental conditions. This will require support for the conduct of field trials that incorporate controlled releases of oil. The consensus of scientists in oil spill countermeasure research is that major advances in the development and validation of spill response technologies are being hampered by our inability to conduct controlled field experiments with oil. Such field experiments involving intentional releases of oil must use scientifically valid and statistically sound experimental designs that use multiple replicate control and treated plots to enable calculation of experimental error.

8.22.1.6 Gap: Sufficient information to conduct NEBA for Arctic spills

Recommendation: Compare the Risks of Various Methods and Intensities of Cleanup under Arctic Conditions (Short-term to Medium-term)

Research comparing sites that have been subjected to various methods and intensities of cleanup to sites with minimal or no cleanup should be conducted under conditions relevant to the Canadian Arctic, building upon the experience in Prince William Sound. Experimental designs should include the range of crude oils that may be produced and transported in the Arctic, as well as the specific fuel oils already transported to Canadian Arctic communities. Well-controlled field experiments would be of great benefit; therefore, the current prohibition of experimental oil spills in the field should be re-examined.

8.22.1.7 Gap: Finer-scale relative risk analysis that allows focused preparedness and response planning

Recommendation: Conduct a Series of Focused Risk Assessments as a Follow-up to Recent Assessments (Short-Term)

The Panel recommends that a set of focused relative risk assessments be conducted building upon the Transport Canada marine spill results by focussing on high-sensitivity areas within each of the assessment zones in order that preparedness and response plans can include explicit plans for the areas with the highest potential consequences.

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Follow-up risk assessments should be focused on specific areas at a finer scale than recent relative risk assessment. In addition, areas can be selected based on environments with multiple anthropogenic and natural stressors and measureable levels of impact and/or potential for future human disturbance. For example, these areas might include portions of the Bay of Fundy, the Gulf of St. Lawrence and the Strait of Georgia. Areas with high natural stressors, or which are sensitive to global factors that, at present, do not have high levels of anthropogenic stressors might include the Labrador Sea, the MacKenzie River delta and Hudson Strait. Once the priority areas for follow-up risk assessments have been identified, methodology refinements should be applied. Recommended refinements are presented in **bold** in Sections 8.11, 8.12 and 8.13.

8.22.1.8 Gap: Limited data from past spills in Canada

Recommendation: Take Advantage of 'Spills of Opportunity' (Short-term and Extending to Long-term)

If spills do occur, maximum advantage should be derived from the opportunity to study the fate, behaviour, and effects of the spill in the short, medium, and long-term. Studies of the relative effectiveness of response measures should also be part of a suite of investigations associated with all significant oil spills in Canada. Pre-approved funding should be put aside for spills of opportunity that would incorporate a combination of research and monitoring. Responsibility for each component of spill studies must be clear and research and monitoring efforts must be well coordinated to eliminate redundancy and to ensure that all required study components are implemented. It may be appropriate to select areas that would not receive treatment; these 'set aside' areas would then serve as a medium and long-term source of data for comparison of treated versus untreated sites.

8.22.2 Medium-priority research needs (medium-term and long-term)

8.22.2.1 Gap: Risks of residual oil in Arctic ecosystems

Recommendation: Evaluate the Consequences of Long-Term Residual Oil in High-Risk Marine Habitats (Medium-term)

The consequences of long-term residual oil in high-risk marine shoreline, estuary, marsh and lagoon areas both north and south of the 60th parallel require further research. Chedabucto Bay is a good candidate for part of this evaluation. Other study areas should be selected on the basis of where consequences might be highest. The results of the research would provide much-needed input to NEBA regarding the most appropriate level of cleanup.

8.22.2.2 Gap: Risks of deposition to deep sea sediments

Recommendation: Evaluate the Consequences of Deposition of Oil to Deep Sea Sediments (Medium-term)

Research into the consequences of deposition of oil to deep sea sediments in the Arctic, as well as deep sea sediments south of the 60th parallel in Canada, would provide part of the required knowledge base for support of decisions regarding offshore oil and gas exploration and oil transport. The current state of knowledge is totally inadequate with respect to supporting confident assessment of risk to deep sea systems.

8.22.2.3 Gap: The potential contribution of hyporheic flow to oil spill risk in rivers

<u>Recommendation:</u> Investigate the Role of Hyporheic Flows in Contributing to Exposure to Oil (Medium-term)

Research is needed on the relative role of hyporheic flows in contributing to exposure of fish to hydrocarbons after a spill. A range of experimental conditions using relevant concentrations and both weathered and unweathered oil should be tested on salmonid species under conditions that represent actual field conditions in terms of variables such as flow, temperature, substrate and dissolved oxygen. Experimental results can be used to evaluate whether monitoring of hyporheic flows after spill events would be feasible and justifiable.

8.22.2.4 Gap: Data supporting assessment of effects on ecosystem services

<u>Recommendation:</u> Investigate the Socioeconomic Impacts of Oil Spills in Support of Assessment of Effects on Ecosystem Services (Medium-term to Long-term)

Measurement of socioeconomic impacts of oil spills as a first step in implementing an ecosystem services approach to oil spill impact assessments. A standard set of indicators of socioeconomic effects should be developed for marine and freshwater systems in Canada, in consultation with First Nations, industry, academics and government agencies. These indicators should have explicit links to ecosystem services. If possible, measurement of effects on culturally important ecological services should be included.

8.22.3 Long-term research needs

8.22.3.1 Gap: Effects of oil spills on trophic dynamics

<u>Recommendation:</u> Conduct Research on the Effects of Oil Spills on Trophic Dynamics of Aquatic Ecosystems

In the Canadian context, the current level of understanding of trophic dynamics in identified high-risk offshore and inshore marine habitats requires evaluation in order that critical gaps in understanding are identified. For example, there may be specific critical features of Bay of Fundy trophic dynamics at specific locations (e.g., estuaries) that, if affected by oil spills, would have much greater consequences than spills in other areas of the Bay. Alternatively, the pelagic trophic dynamics of the Bay of Fundy are fundamental to maintenance of many highly-valued ecosystem services. Currently, it would be extremely difficult to differentiate risks among inshore and offshore areas (and thus to guide cleanup and follow-up monitoring programs).

8.22.3.2 Gap: Potential for population-level effects

<u>Recommendation:</u> Conduct Long-term Research on Effects of Different Oils on Populations of Aquatic Biota

The Panel recommends that there be support for long-term research into effects of different oil types on populations of aquatic biota, especially fish, marine mammals, and waterfowl. Such long-term research requires stable funding and effective collaboration among disciplines such as environmental chemistry, toxicology and ecology. Spills of Opportunity could provide the impetus for the initiation of long-term research.

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8.22.4 Operational preparedness needs

Standardized Sensitivity Mapping

National-scale relative risk assessments used for preparedness and response planning must be based upon reliable and consistent data and analysis; therefore, standardized methodology for environmental sensitivity mapping is an *urgent requirement*.

Post-Spill Monitoring Program Development and National Guidance

The Panel strongly recommends that national guidance regarding monitoring after oil spill events be developed, involving consultations among industry, government, Indigenous communities and community stakeholders. The guidance would be designed to produce information that is reliable, adequate, credible and consistent. The guidance should include provisions for adjustment in response to specific characteristics of the receiving environment. The guidance should also include requirements for standard baseline datasets (tailored to specific receiving environments). The guidance could be divided into two parts: 1) which information to collect without exception; and 2) which information gathering can be deferred until the full scope of the spill and its potential effects are better understood. The guidance should include information on data quality requirements, including determination of minimum sample sizes, standard sampling protocols and laboratory quality assurance/quality control.

The Panel recommends the development of a national, consensus-based set of indicator species for each of Canada's major marine offshore and inshore zones (perhaps using the zones developed by WSP and SL Ross (2014). Once these indicator species have been selected, coordinated research programs among academia, industry and government should be developed which: a) compile existing information; b) identify and prioritize critical data gaps; and c) conduct research to fill the priority data gaps.

Standard guidance for monitoring the fate of spilled oil in all major environmental compartments of lotic systems is required. This guidance would include measurement of oil in sediments, as noted above, but would extend to measurement in water, pore water and biota.

Standard guidance is required for measurement of oil contamination of sediments in flowing waters in Canada. This guidance should consider the range of lotic receiving environments and logistic constraints common to remote locations. The guidance should include the de minimis level of investigation required for decision-making regarding cleanup requirements and techniques, as well as for determination of acceptable residual levels of oil.

The Panel supports the recommendations of SL Ross (2014) regarding monitoring programs in Canadian aquatic systems. These recommendations are listed in Section 8.14.

Update and Refine Risk Assessment Methods for Use in Support of Project Applications or Preparedness Planning

The Panel's review of risk assessments performed as part of pipeline applications revealed opportunities for improvement.

- Risk assessment scenarios should assume a more likely combination of factors (less than full-bore rupture with a longer elapsed time to response) that are more relevant and comparable to actual spill incidents. These scenarios would also be viewed as more credible by stakeholders sensitized by recent spills that experienced long delays before shutdown.
- Detailed risk assessments conducted in support of applications for new projects such as pipelines and production facilities should include winter conditions.
- Stochastic fate modeling for complex, sensitive receiving environments is recommended.

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- The sites of interest used in the EEPP assessment merit being re-analysed using more sophisticated analytical tools and approaches—particularly with respect to fate and transport and the level of sophistication with respect to acute and chronic toxicity, shoreline effects and overland flow effects with season.

Apply Research Results, Engagement with Indigenous peoples and Stakeholders and Economic Analyses to Address the Longstanding Remediation Question "How Clean is Clean Enough?"

Effects on ecosystem services can be evaluated using data obtained from laboratory studies, large-scale open-air experiments, long-term field research and baseline data surveys. These data should be compiled and interpreted with the intention of determining acceptable, science-based chemical and biological endpoints of spill remediation. Long-term monitoring of follow-on effects after experimental spills and spills of opportunity would inform regulatory guidelines for cleanup endpoints.

The benchmarks for an acceptable level of residual oil should always include social and economic factors. Social factors must include input from Indigenous peoples. Social and economic factors could include residual effects on ecosystem services such as recreational or commercial fishing, drinking water sources and parks or other protected areas.

Inclusive Preparedness Planning

Preparedness planning should consider information from Indigenous, provincial, territorial and municipal sources in order that smaller-scale, higher-risk locations can be identified. Many of these areas have already been mapped at a finer scale than presented in the Transport Canada relative risk assessment. Provincial laws and regulations must be considered in preparedness to ensure harmonization and avoidance of redundancy.

Develop a Spill Risk Calculator Tool

The Panel suggests that a Spill Risk Calculator tool for Canada would be beneficial to preparedness planners and responders. This tool can be based upon methods used for the assessment of oil spill risks in Alaska (Reich et al. 2014).

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CHAPTER 9: SUMMARY CONCLUSIONS AND RECOMMENDATIONS

The Royal Society of Canada (RSC) Expert Panel on The Behaviour and Environmental Impacts of Crude Oil Released into Aqueous Environments was asked to address the following questions:

1. How do the various types of crude oils, including diluted bitumens, compare in the way they behave when mixed with fresh, brackish or salt waters under a range of environmental conditions?
2. How do the various crude oils compare in their chemical composition and toxicity to organisms in aquatic ecosystems?
3. How do microbial processes affect crude oils in aquatic ecosystems, thereby modifying their physical and chemical properties, persistence and toxicity?
4. Are the research and oil spill response communities able to relate, with reliable predictions, the chemical, physical and biological properties of crudes to their behaviour, persistence, toxicity and ability to be remediated in water and sediments?
5. How should these scientific insights be used to inform optimal strategies for spill preparedness, spill response and environmental remediation?
6. Given the current state of the science, what are the priorities for research investments?

During its deliberations the Panel considered accidental releases of conventional (light, medium and heavy) crude oils and unconventional crude oils (including diluted bitumen), at exploration and production sources or as they are transported between ports and storage sites, to refineries or to users.

Following a review of available information, the Panel concluded that large spills of crude oil into marine or freshwater systems in Canada from oil production facilities, tankers, pipelines, rail and truck transport that cause significant ecological damage are infrequent, and the probability of spills decreases with increasing spill size (Chapter 1). The Panel noted that comprehensive statistical records for spills into the marine environment of Canada were much more readily available than for inland spills into fresh water. Thus, in the case of freshwater spills, the Panel had to search through individual investigative reports from the Transportation Safety Board (TSB), Alberta Energy Regulator (AER) and the media. Furthermore, as illustrated by the AER (2013) report on pipeline performance that provided data on spill incidence and volume, spills of crude oil were not always distinguished from refined product spills. Instead, statistics were presented as 'hydrocarbon liquids'. This disparity in spill record keeping for the freshwater environment of Canada should be resolved.

Despite the relative infrequency of crude oil spills into aqueous environments in Canada, the consequences of spills into sensitive waterbodies can be substantial, not only economically but also impacting human health, safety and the environment. This concern is commensurate with the key questions posed by the sponsors and stakeholders.

With the anticipated growth of Canada's offshore oil and gas industry and the production and transport of unconventional oils, such as diluted bitumen, accidental releases of oil from offshore platforms, pipelines and other sources were considered to be within the scope of this report. The Panel focused on the Canadian environment. However, it reviewed and considered the applicability of case studies from the United States and other countries to ensure full coverage of pertinent information. For example, current knowledge on the effects of and response to subsurface blowouts was provided by the Gulf of Mexico Deepwater Horizon (DWH) oil spill, and the consequences of and response to a release of dilbit spilled into a freshwater environment was represented by the Kalamazoo River, MI, oil spill. The Panel has made an effort to acknowledge the large number of research papers and environmental impact statements recently published on oil spills, as well as major ongoing research initiatives. The conclusions and recommendations made by the RSC Expert Panel in this report, under its remit to address the questions of its Terms of Reference, are subject to revision as new findings from environmental impact assessments and oil spill response technologies emerge.

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Question 1: How do the various types of crude oils, including diluted bitumens, compare in the way they behave when mixed with fresh, brackish, or salt waters under a range of environmental conditions?

The Panel provided an overview on the chemical composition of petroleum hydrocarbons, the types of conventional and unconventional oil (including dilbit) transported in Canada, their bulk properties, and information on the processes that influence their weathering in the environment (Chapter 2). The Panel concluded that the behaviour of these oils when mixed with fresh, brackish or salt waters is dependent on their inherent physical and chemical properties, which are influenced by the environmental conditions at the specific areas of concern. Our understanding and ability to accurately predict the behaviour of some oils in the environment following accidental spills are still limited. This is due to analytical challenges, incomplete experimentation and complex interactions among oil weathering processes, remediation strategies and environmental conditions.

Each oil spill will be a unique combination of oil and receiving environment characteristics. However, the review of oil spill cases revealed that some generalizations can be made (some having more uncertainty associated with them than others).

Season and weather are dominant factors determining both the initial and ultimate fate and effect of all oil types. The lighter the oil, the more it is affected by spreading and evaporation and the easier it is to treat effectively, regardless of the environment in which it is spilled. The higher the ambient temperature, the faster is the evaporation and dissolution and the more extensive is the spreading. Medium oils (e.g., most conventional crude oils) are generally less volatile than lighter oils, which increases the potential for impacts on waterfowl and wildlife. However, medium and light oils are amenable to relatively rapid biodegradation. Heavy oils and diluted bitumens (e.g., Bunker C, dilbit, synbit and railbit), which have fewer components that dissolve in water or evaporate to the atmosphere are more resistant to evaporation and biodegradation and thus their potential long-term damage to the environment, waterfowl and fur-bearing animals is greater. Cleanup of heavy oils and bitumens is extremely difficult for both marine and inland spills because of their specific gravity, viscosity, flash point properties and high asphaltene content.

In addition, the direct effect of weather conditions on turbulence influences the physical dispersion of oil and the likelihood of successful dispersant use, especially for light and medium oils where some degree of mixing-energy is required for breakup of oil into small droplets. However, turbulence can also increase emulsification of oils, especially heavier oils, causing moussification or the production of water-in-oil emulsions, which resist evaporation, spreading and biodegradation. Dilbit is an exception because it resists the formation of water-in-oil emulsions. Turbulence is also an important factor in the formation of oil-particle aggregates (OPAs; significant with respect to some diluted bitumens and their potential for sinking).

High winds with associated wave action and currents can deposit oil onto windward shorelines and into more protected areas such as lagoons. High waves, tides and seiches can result in oil burial along sandy shorelines followed by re-exposure. Wind-driven surges can drive oil into the high intertidal zone or, in freshwater systems, deep into marginal wetlands or riparian habitats. Spills occurring immediately after high rainfall events can transport and subsequently deposit oil over wide areas of the floodplain of rivers, resulting in significant stranding when flows subside. These effects are applicable to all oil types, regardless of their properties.

The persistence of residual oil along shorelines correlates with shoreline energy level (e.g., waves, tides and ice), with oil persisting longer in low-energy and low-temperature environments. Cobble/boulder beaches with fine-grained interstitial materials, lagoons with fine sediments, backwater channels and shoreline marshes have been linked to the sequestration of conventional crude oil in a relatively un-weathered state for long periods of time due to protection from physical weathering and/or biodegradation in hypoxic conditions. Heavier oils are less amenable to this sequestration action as they tend to remain on the surface and resist seeping into the interstitial area of the shoreline.

The fate of oil spilled into rivers is affected by river flow, river channel characteristics and sediment characteristics and processes. High-flow conditions at the time of the spill will increase turbulence and the presence of suspended particulate matter that, in turn, contribute to the formation of OPAs that will sink and accumulate in deposition zones where they may be re-suspended by subsequent high-flow events. Biological processes within sediments can also elicit the re-suspension of oil and associated sheen production on the water surface. River channels that have been subject to bank hardening (e.g., by placement of riprap or rubble) have higher velocity flow, leading to the potential for transport of spilled oil farther downstream and/or stranding of oil among the bank materials. There is also a potential for entrainment of oil into porous bed sediments of shallow, high gradient rivers by surface-ground water interactions (hyporheic flows), although this possibility has not yet been evaluated in detail.

The fate of oil spilled into lakes is affected by lake morphology and mixing characteristics. Conventional crude oils spilled into shallow, wind-swept, well-mixed lakes will be subject to rapid dispersion if the spill occurs during open-water seasons. Oil, especially the heavier types, spilled into deeper lakes with thermal stratification may, in part, be sequestered in cold, low-oxygen, deepwater sediments with subsequent low biodegradation rates if OPAs are formed and subsequently sink.

Brackish and saline waters have greater density than fresh water, which affects the sinking of oil. In theory, as a dilbit loses its light components to evaporation, it may reach sufficient density to sink even in seawater, especially if it forms OPAs. The longer-term fate and effects of oil deposits within marine sediments is the subject of current research.

Question 2: How do the various crude oils compare in their chemical composition and toxicity to organisms in aquatic ecosystems?

The Panel provided an overview of what is known about oil toxicity and ecological impacts of oil spills to aquatic environments based on previous review articles and current literature (Chapter 4). Numerous case studies (e.g., *Exxon Valdez*) have documented various levels of habitat destruction that have resulted from oil spills. Although an overview is provided on the toxic effects of oil spills on various biota (i.e., plants, invertebrates, fish, reptiles/amphibians, birds and mammals) within the Panel's report, a major focus was given to fish, which were the subject of the greatest number of research papers. This is not surprising as the concern over oil spills is largely focused on impacts to fisheries due to their socioeconomic importance. Furthermore, many of the interactions between oil chemistry and its toxicity to fish may also apply to other species.

Studies with fish have shown that acute lethality is associated primarily with low molecular weight (LMW) components of oil, such as monoaromatics (e.g., BTEX), diaromatics (naphthalene and alkyl naphthalenes) and short-chain alkanes ($< C_{12}$). LMW petroleum hydrocarbons are sufficiently hydrophobic to partition rapidly into lipid membranes and to cause mortality by narcosis. Typically, as oil weathers following a spill, most narcotics hydrocarbons are lost within two days by volatilization, biodegradation and dilution. As a consequence, lethal concentrations in water are not maintained for prolonged periods unless the discharge of oil is continuous. Thus, observed fish kills are typically brief and localized because of the rapid loss of the acutely lethal LMW oil components through dilution and weathering. However, extensive fish mortalities have been observed in rivers where a point source of oil was rapidly transported downstream before significant weathering occurred (e.g., pipeline break, Pine River, BC).

The acute toxicity of hydrocarbons to aquatic biota can be estimated from octanol-water partition coefficients (K_{ow}), the ratio at equilibrium of the concentrations of a compound in n-octanol (a water-insoluble phase that mimics tissue lipids and floats on water) to the concentrations in the underlying water. Mortality typically occurs when tissue concentrations reach a 'critical body residue' or median lethal dose (CBR or LD50) of approximately 5 mmoles/L of tissue lipid. These predictable relationships have been used as the basis for mathematical models for assessing the risk of acute lethality following potential oil spills from proposed developments, such as pipelines, and for assessing the potential lethality of hydrocarbon mixtures measured during actual oil spills.

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Chronic and sublethal effects of oil on fish are associated with either chronic exposures (e.g., oil stranded in sediments) or with delayed or lingering effects of acute exposures. Over the lifetime of a species, exposures to oil may occur via respiratory uptake o f hydrocarbons from water (most common), direct contact with oil droplets (an emerging area of research) or the diet and maternal transfer to eggs (not well studied). Effects range from initial cellular genetic and molecular responses to long -term organismal impacts on rates of reproduction, growth, disease, mutations/cancer and survival.

In contrast to acute toxicity , 3-5 ring medium and high molecular weight (MMW and HMW, respectively) polycyclic aromatic hydrocarbons (PAHs), including aromatic heterocycles, such as alkyl dibenzothiophenes, are thought to be primary causes of chronic toxicity to fish. Embryos appear to be the fish life stage most sensitive to oil exposure, presumably due to interference with the large number of genes that must be up-regulated or a ctivated at precise stages during embryonic development for normal development to occur. Most studies report the effects of waterborne oil, usually by reference to the concentrations of TPAHs and total petroleum hydrocarbons (TPHs) in test solutions that a ffect 50% of test organisms. The EC50 and LC50 values for chronic exposures have been reported to range from 0.3 –60 µg/L TPAH and 0.03 – 11 mg/L TPH. Because of their lower volatility, alkyl PAHs persist in water longer than monoaromatics continuing to partition from oil into water over weeks to months if residual oil persists.

It might be expected that the relative risks of acute and chronic toxicity of different oils could be predicted by simply ranking them in order of the concentrations of their di fferent constituents. Unfortunately, the expected correlations were not observed in studies with marine fish and invertebrates. Toxicity may be confounded by the relative proportions of different alkyl PAH s, the presence of unidentified compounds (includin g additives) or by generation of polar hydrocarbon derivatives during weathering of the oil.

Differences in the relative proportions of different PAHs in oil may influence the toxicity of diluted bitumens, such as the product dilbit . Dilbits are prepared from bitumen, a highly -weathered oil depleted in alkylnaphthalenes, which are the alkyl PAH of lowest chronic toxicity. Thus, dilbit contains a higher proportion of the more toxic 3 - to -5-ring alkyl PAH mixture than do conventional crude oils, s o that chronic toxicity at a given concentration of TPAH may be greater than expected. Only one report has been published on the chronic embryo toxicity of dilbit. In this study with Japanese medaka, chronic toxicity across an array of crude and refined oi ls suggested that dilbit may be slightly more embryo -toxic than average. Until equivalent data for other species have been reported, any conclusions about the perceived toxicity of dilbit may be confounded by species differences in sensitivity to oil. There is some evidence dilbit elicits acute toxicity in benthic invertebrates and that its toxicity varies with sediment characteristics.

It is often difficult to express quantitative relationships between PAH concentrations in oil and toxicity because the physical characteristics of oil modify toxicity by controlling droplet formation and the rate of partitioning of hydrocarbons from oil to water. The proportion of waxes, resins and asphaltenes will control the physical characteristics of oil, which in tur n affect the dispersability and biodegradability of oil in water. Thus, relative to light oils, heavy oils including dilbit are more difficult to disperse mechanically in water, are less likely to form fine droplets, biodegrade more slowly, and may appear less toxic than light oils because their toxic constituents are less bioavailable.

The Panel's report includes tables and references that contain compositional and physical data for crude oils that represent petroleum types transported in Canada and global ly. Based on current analytical capabilities, the petroleum hydrocarbons produced and/or transported within Canada appear to fall along a chemical continuum spanning condensates to bitumen. Condensates are considered acutely toxic due in part to elevated B TEX concentrations. Conventional oils with lower proportions of BTEX can exhibit chronic toxicity associated with alkyl PAHs and resins.

The results of studies to date suggest that the direct toxic effects of dilbit in both freshwater and marine ecosystems do not seem to differ dramatically from other oils. However, the binary nature of

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dilbit currently confounds extrapolation of toxicity from its individual components: diluent and bitumen. The acute toxicity associated with the diluent may be dispelled by evaporation, and the heaviest residues (the asphaltenes) in the bitumen are likely biologically inert. However, the likely extent of diluent evaporation (conversely, the biological availability of diluent in weathered dilbit) is currently unknown. Furthermore, the toxicity of resins and/or products of photo oxidation is poorly understood.

The impacts of spilled oil on aquatic species depend as much on the extent of their exposure to oil as on the inherent toxicity of oil. Exposure to oil will vary with the interactions among the fate and behaviour of spilled oil and the habitat characteristics, behaviour and physiology of different species. Research is needed to identify those species that will be most exposed to oil with extreme properties, including very light (e.g., Bakken crude) and very heavy oils (e.g., HFO), and the ecological factors that will enhance exposure (e.g., turbulent mixing of rivers or weathering). Research is also needed to better understand the differences between dilbit and Bakken light crude versus conventional crude oils in terms of their acute and chronic toxicity. The hydrocarbon composition of various oils can play a major role in explaining these differences. Although acute lethality can be attributed to LMW components of oil and chronic toxicity to the alkyl-substituted 3- to 5-ring PAHs, research is needed to identify the most toxic constituents of oil, particularly polar hydrocarbons generated by weathering, biodegradation or photooxidation that are not often analyzed in oil or water.

It is important to note that most vertebrate species (e.g., fish, birds, mammals) can metabolize and excrete hydrocarbons. In contrast, invertebrate filter feeders do not metabolize or excrete petroleum hydrocarbons quickly; thus, contaminated zooplankton can accumulate significant quantities of droplet oil. Although contaminated invertebrates can contribute to the dietary exposure of predators, it appears that petroleum hydrocarbons do not typically biomagnify to high concentrations in food webs.

Question 3: How do microbial processes affect crude oils in aquatic ecosystems, thereby modifying their physical and chemical properties, persistence and toxicity?

The Report outlines in Chapters 2, 3 and 6 how microbial communities in aquatic environments interact with spilled oil to change its properties, behaviour and effects on other biota. Interactions that affect physical properties include stabilizing emulsions via cell attachment to oil:water interfaces, enhancing oil dispersion in water through secretion of biosurfactants and biodegradation.

The chemical composition of oil dictates its susceptibility to biodegradation (i.e., the proportions of biodegradable alkanes and aromatics versus recalcitrant steroids, resins and asphaltenes). In competition with abiotic evaporation and dissolution processes, biodegradation selectively converts certain lighter alkanes and aromatics to gases (CO_2 and/or methane), water and biomass while leaving the heavier, complex oil components as residues. This altered chemical composition changes the physical properties of the residual oil, making it more dense and viscous than the original. In turn, this physical change progressively reduces further biodegradation of the residual oil since biodegradation is also influenced by the physical state of the oil (e.g., viscosity that affects the oil:water interfacial surface area available for microbial attachment and attack). Biodegradation, being an enzymatic process, is directly affected by temperature, and because microbial cell growth is the ultimate outcome of biodegradation, its rate is affected by availability of growth-promoting nutrients such as soluble nitrogen and phosphorus sources. Even under ideal environmental conditions, biodegradation of the multitude of components within a crude oil is a relatively slow process that may take weeks, months or even years to remove a significant mass of spilled oil, and oil that is buried in anaerobic sediments may persist for decades or longer. A residue of original petroleum components, including asphaltenes and resins, will always remain, in addition to partially degraded hydrocarbons and metabolic end products (e.g., CO_2, CH_4, H_2S), the mass of which will depend on the interplay between environmental conditions, petroleum chemistry and microbial community composition.

The predominance of aerobic versus anaerobic conditions dictates which enzymatic pathways are effective, the rate of microbial attack and the end products. Oil biodegradation is faster and more efficient under aerobic conditions, such as in a well-aerated water column or in the uppermost layer of

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sediment, but it is slower in environments dominated by anaerobic conditions, such as buried sediments. The range of susceptible hydrocarbons under anaerobic conditions appears to be more limited than with aerobic degradation, and the end products and residual oil composition therefore differ from aerobic processes. For this reason, wetlands, salt marshes and mud flats that have significant anaerobic strata are among the most sensitive and difficult ecosystems to clean after an oil spill. Bioremediation may be the best approach for wetland cleanup because it is among the least intrusive strategies for these sensitive environments. However, even bioremediation requires some human intervention that may cause additional harm to the contaminated site by trampling of oil into the anaerobic sediment, which may diminish bioremediation of the site and promote oil persistence *in situ*. Natural attenuation may be the cleanup method of choice if trampling cannot be avoided. The beneficial interaction of hydrocarbon-degrading microbes and plant roots in soils and wetlands (phytoremediation) may enhance biodegradation and decrease persistence of buried oil.

The types of microbes present at a spill site dictate the lag time before biodegradation begins, the subsequent rate and extent of biodegradation and hence the behaviour and persistence of the residual oil. Hydrocarbon-degrading bacteria are ubiquitous, but they tend to be present in higher proportions in chronically oil-impacted sites because they have an advantage over microbes that cannot utilize hydrocarbons. For the same reason hydrocarbon-degraders may flourish after an oil spill even in a pristine site, given favorable conditions. Thus, the history of a spill site may indirectly influence progression of bioremediation or, conversely, oil persistence. Extensive and rapid rate of biodegradation requires the metabolism of a consortium of hydrocarbon degraders working in conjunction with non-hydrocarbon degrading microbes.

Given these general factors, it is evident that light oils are inherently more biodegradable than heavier oils and leave lighter residues. In contrast, only small proportions of heavy crudes are readily biodegradable, and the residues will persist even with active intervention. In the case of dilbit,it is not yet clear whether all of the diluent is actually available for biodegradation, is protected by virtue of molecular association with the bitumen or is simply unavailable to microbes because of the limited surface area of viscous weathered dilbit. Considerable research is required to address this knowledge gap under different environmental conditions.

In the process of selectively biodegrading petroleum components, microbes can reduce or increase the toxicity of an oil spill. For example, very light oils (e.g., jet fuels and gasolines) contain high concentrations of acutely toxic compounds that are susceptible to biodegradation. Although such light compounds may be quickly lost by abiotic processes (particularly evaporation and dissolution), biodegradation may reduce acute toxicity if they are retained in an environment (e.g., in deep water or buried in sediment or trapped under ice). In some conventional oil spill scenarios natural attenuation can effectively lower the risks to aquatic life and human consumers of fish and shellfish, despite persistent residues, because: a) biodegradation may produce primarily non-toxic residues; b) the residual oil is not being released back into the water in amounts sufficient to significantly affect aquatic life; or c) residual oil compounds (e.g. asphaltenes) are not bioavailable. This was well illustrated by the *Arrow* tanker oil spill case study. Microbial degradation by indigenous bacteria (i.e., natural attenuation) contributed to the reduction of toxic fractions within the heavy Bunker C fuel oil stranded within the beach sediments at some sites. At present, information is insufficient regarding the effectiveness of monitored natural attenuation as an oil spill countermeasure for a dilbit spill in the aquatic environment.

Conversely, in unfavorable conditions microbes may be unable to completely convert hydrocarbons to gases, water and biomass. Instead, partially oxidized metabolites that are more polar (more water-soluble) and more mobile in an aqueous system may accumulate. Because of the diverse hydrocarbon isomers present in petroleum, an enormous number of possible metabolites could be produced. The toxicity of most is untested either as pure compounds or, more realistically, as mixtures of polar metabolites. In some cases the metabolites generated by microbial attack are known to have carcinogenic potential. By analogy to the resins fraction, many are likely to have chronic toxic effects. Additional research is needed to address this question of chronic toxicity due to incomplete microbial metabolism of crude oil components.

Another factor affecting chronic toxicity of the oil itself (versus its metabolic products) is that as lighter components of the oil are biodegraded, the proportion of alkyl PAHs and resins in the residual oil increases, thus increasing the chronic toxicity associated with these component s. In addition to these chemical effects, the increase in oil density and viscosity due to biodegradation of light fractions can increase the likelihood of detrimental physical impacts on higher organisms including plants.

The mechanism by which microbial processes affect dilbit toxicity has yet to be fully resolved because the binary nature of dilbit confounds simple extrapolation of toxicity from the individual components.

Question 4: Is the research and oil spill response community able to relate, with r eliable predictions, the chemical, physical and biological properties of crudes to their behaviour, persistence, toxicity and ability to be remediated in water and sediments?

The environmental behaviour of unconventional oils and blended oils currently can not be predicted with confidence, which affects spill response planning and decisions. Each oil spill is unique, due to the specific chemical nature of the oil and the complex interaction with different environmental and climate factors. Thus, to account for the environmental variables that might be experienced in a dilbit spill, a multitude of combinatorial trials would be required before empirical rules of its behaviour c an be developed.

The toxic effects of oil are well described for a limited array of aquatic organisms, most notably acute toxicity in fish embryos due to their high sensitivity, inability to avoid exposure and the obvious concerns over the impacts of oil spills on the fisheries. Much less is known about the effects of oil on other responses of fish (e.g., sexual maturation, reproduction, behaviour) or on other species, in part because research has not been focused and systematic (e.g., see review of toxicity to algae, Chapter 4) or it is simply lacking. In addition, the interactions of oil exposure and toxicity with environmental stressors, including temperature, salinity, oxygen concentrations, pH, spawning migrations and other anthropogenic contaminants, would be important variables in this research. It is critically important to determine if acute sublethal exposure of aquatic organisms (e.g., fish embryos) to oil causes delayed or lingering toxicity evidenced as a reduced abundance in the years following an oil spill.

A major impediment to quantitative predictions of oil toxicity is the quality of toxicity data for risk and impact assessments. Problems include a lack of methods to distinguish dissolved from droplet oil, the high cost of analyt ical methods to adequately characterize exposures, and time -varying concentrations in toxicity tests. The application of toxicity data in ecological risk assessments is also limited by the availability of models capable of linking acute and chronic toxicit y to the chemical characteristics of hydrocarbons. The current models that predict acute lethality reasonably well do not apply to situations where concentrations of oil in water vary over time, as would be the case during an oil spill. The routine applica tions of these models also do not generate estimates of the error of calculated toxicities, despite considerable variations between observed and predicted toxicities for individual compounds. Models to predict chronic toxicity have not been developed from data on chronic toxicity, but rely on the incorrect assumption of a constant ratio between acute and chronic toxicity.

Ecological impact assessments of spilled oil, especially dilbit, are also limited by a lack of ecological research in oil-contaminated ecosystems. Long-term time-series research ranging from days to years on 'spills of opportunity' or experimental oil spills is needed to establish mechanistic and statistical links among hydrocarbon exposure of aquatic species, biomarker responses, the prevalence of signs of toxicity and pathology, the distribution, abundance and production of individual species, and ecosystem structure and function. In particular, research is needed on the fate, behaviour and effects of oil in freshwater ecosystems usi ng designs and methods that reflect differences in spatial scales, processes, water movements and biological communities relative to marine ecosystems.

A knowledge gap remains in understanding the potential impacts of oil, or chemically -dispersed oil, in deepwater and Arctic ecosystems where dispersant use is contemplated. The complex interactions

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among extreme low temperatures, the unique physiologies of coldwater species, the solubility kinetics of hydrocarbons in water, exposure times, pharmacokinetics of hydrocarbon uptake and toxicokinetics for different mechanisms of acute and chronic toxicity of oil must be investigated. With respect to freshwater and terrestrial ecosystems, our understanding of the impacts of spilled oil on permafrost ecosystems and northern wetlands, which are characterized by complex water flows and plant and animal species highly adapted to harsh conditions, is severely lacking.

Chronic impacts may also result from a cascade of effects originating with acute exposure to the spill and/or to lingering effects of residual oil. For example, mortality induced damage to the structure of a population or community (e.g., a killer whale pod) will alter food web dynamics (e.g., elimination of a predator leading to increased abundance of the prey species and increased feeding pressure on primary producers). Toxicity can also cause indirect effects, such as energetic costs associated with the metabolism of PAHs and other hydrocarbon compounds, that result in lower body weight and increased mortality or higher susceptibility to predation. Such effects and their ecological significance are not fully understood and hence not properly considered in risk assessment models.

In terms of modeling, the oil spills of interest for the community are those that cover large areas because the smaller ones can be more or less contained and treated. Therefore, the thrust of modeling oil transport from a response point of view has been on parametrizing small-scale processes while attempting to capture the bulk of the large-scale transport. In other words, the small-scale information has been sacrificed in favour of the large-scale. Unfortunately, this means that most oil spill models designed for response, such as NOAA's GNOME, are not designed to provide concentrations that could be readily used for predicting toxicity effects. For example, a model that conserves mass and uses a computational block of 1 km x 1 km would not distinguish whether the mass of oil is spread within that block or whether it is localized within a 10 m x 10 m sector of that block. While the mass is the same in both cases, the latter provides a concentration that is 100 times greater. Also, oil spill models developed for response cannot be used to design the optimal parameters for applying dispersants. In contrast, models designed for research are sufficiently sophisticated to allow their use in toxicity studies and for evaluating remediation scenarios. Nevertheless, as modeling is becoming more acceptable as a predictive tool, sophisticated models are becoming more common. The goal is to achieve effective communication between the response community and the academic community that can lead to collaboration, such as occurred with the droplet model from jets VDROP-J developed in academia and currently being adopted by NOAA in their model ADIOS2.

To support the application and accuracy of predictive models following spill events, research and implementation of autonomous, remotely operated detection devices (e.g., AUVs and ROVs) and other remote sensing technologies should be encouraged, as well as the development of new or refined biomarkers and fingerprinting methods for conventional and unconventional oils (including dilbit) and oil spill treatment agents such as dispersants, in water, sediment and biological materials recovered from freshwater and maritime systems. Uncertainty, in the form of incomplete information, errors in sampling, subjective judgement, random variations of and dynamic interactions among operating factors, approximations and assumptions in measurement, and changes of environmental conditions, will always be a challenge in oil spill response operations. Reduction in the level of uncertainty is needed in the measurement of environmental factors, spill modeling and response decision-making and implementation.

The Panel recommends the conduct of biodegradability studies on conventional and unconventional oils commonly transported in Canada, especially dilbit, to delineate the maximum biodegradation rates expected under actual environmental conditions. Comparison to multiple control conditions will be essential for rigorous interpretation of the results. A controversy still remains over the use of oil dispersants for spill response. Consideration should be given to the use of mathematical decision-making tools to optimize the dose, logistical and operational processes for their application, as well as the development of more effective and eco-friendly dispersant formulations.

The existing models for simulating response processes (e.g., booming, *in situ* burning, skimming, dispersion and bioremediation) and predicting their effectiveness (individually and collectively) are

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limited in terms of accuracy and adaptability to different environmental conditions. This is especially true for cold waters and Arctic environments. Furthermore, once the response has ended, it becomes a matter of evaluating the effectiveness of natural attenuation (i.e., a reduction in contaminant levels and habitat recovery) and risk from residual oil. The development of standardized methodologies and targeted post-spill monitoring programs are needed to accurately determine the levels of response efficacy and the risks of residual oil.

The challenge exists on how to determine what level of preparedness is appropriate for each area of response and of concerns across Canada. Despite the importance of oil type, the Panel concluded that the overall impact of an oil spill, including the effectiveness of an oil spill response, depends mainly on the environment and conditions (weather, waves, etc.) where the spill takes place and the time lost before remedial operations.

In terms of prevention, there is a need to assess the current level of preparedness in light of relative risks in order to identify major geographic areas of concern and those with deficiencies in response capability. The risk assessment commissioned by Transport Canada should be expanded from marine spills to freshwater environments. In addition to cumulative impacts of response actions, socioeconomic factors should be more effectively reflected in the net environmental benefit analysis (NEBA) process to minimize impacts on a region's environment and economy and the wellbeing of its people. In terms of risk assessments, it is important to stress that our ability to distinguish the effects of oil spills (even on the scale of the DWH) depends upon the availability of pre-spill baseline information, an understanding of natural variability and an understanding of the effects of multiple anthropogenic stressors. The risks of oil spills must also be evaluated in the context of large-scale oceanic and climatic phenomena and changes that are occurring at unprecedented rates.

In the marine environment, preliminary data suggest that the early post-spill behaviour of dilbit would not differ substantially from the behaviour of conventional crude oil. However, dilbit fate and behaviour depends on various and sometimes unpredictable environmental factors, such as the level of mixing energy and level of contact with suspended particles. Thus, questions remain about our level of confidence in selecting and developing suitable response strategies and cleanup equipment for both contingency planning and response management. To reduce the level of uncertainty, rapid and effective response operations are paramount. Regardless of oil type, rapid response will limit the spatial extent of floating oil, prevent extensive contact with soils, sediments and organic materials along the spill flow path, and minimize shoreline stranding. The nature of the crude oil is less important than the rapidity and efficacy of the response. For example, rapid on-water response by an oil recovery unit to the spill of diluted bitumen from the punctured Kinder Morgan pipeline in Burnaby, BC, minimized subsequent environmental damage. However, if such a spill had occurred in a more remote area, the consequences might have been more substantial.

Cold and harsh environments in northern Canada and the Arctic are usually characterized by a wide range of wind speed and direction, limited visibility, low temperature, rough water surface, ice coverage, etc., posing substantial challenges for oil spill prevention, preparedness and response. To protect the habitat and its resources within the Arctic and sub-Arctic regions, vulnerability and risk analyses are needed to support contingency planning and response operations. Currently, there is a drive within Canada and internationally (e.g., Northern Region Persistent Organic Pollution Control Laboratory – Memorial University; Churchill Marine Observatory – University of Manitoba; Arctic Joint Industry Partnership – International Association of Petroleum Producers) to increase understanding of the fate and behaviour of oil in water and ice conditions (solid, slush and frazil ice) and during active periods of formation and breakup of annual and multi-year ice, as well as the impacts on ecosystems. This experimental information on the effect and efficiency of different response methods under Arctic conditions is urgently needed to improve the ability to forecast and manage oil spill response operations in a more timely, eco-friendly and cost-effective manner.

Most of the knowledge of oil spill response technologies has been developed through results from studies in the laboratory, mesocosm test systems and actual case studies (with limited controls and treatment replication). It is the consensus of the Panel and general scientific community that

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controlled field trials are needed to advance the development, validation and acceptance of operational spill response strategies, especially for subsurface blowouts, Arctic oil spills and freshwater shorelines.

Question 5. How should these scientific insights be used to inform optimal strategies for spill preparedness, spill response and environmental remediation?

Baseline data are needed to distinguish oil impacts from natural environmental variability and for the selection of spill response options to be taken (e.g., dispersant use). The identification of operational endpoints for oil spill response operations raises the difficult y of addressing the question "how clean is clean?" Usually this is a site-specific question because habitat recovery depends largely on a variety of species at risk identified through baseline surveys.

Anticipating the challenges of oil spill response in remote, northern areas and cold waters and planning for each c ontingency would help decrease the likelihood of undue delays and inappropriate response techniques. Recovery of oil spilled in the Arctic can be expected to be relatively ineffective; therefore, prevention must remain a major focus.

Delays in response typify oil spill case studies. Common causes of delays include lack of preparedness and response capabilities, deficiencies in regulatory oversight and lack of communication and coordination among government agencies, Indigenous communities and industry. Involvement of Indigenous communities in decision -making regarding cleanup and monitoring is occurring and will continue to contribute to the baseline knowledge required to support focused preparedness planning.

Human intervention often can cause more harm than the spill itself. Aggressive shoreline cleanup methods, such as high-pressure washing and physical removal of oiled vegetation and debris along the shore of lakes and rivers, can impart substantial negative effects on biological communities, including indirect and cascade effects that can delay recovery. Thus, responders should be sure to consider the impacts of their response actions will have on the ecosystem even from the contingency planning stage. In some cases, monitored natural attenuation (natural recovery) may be the best response option for stranded or submerged oil since the negative effects of cleanup may be significant. Consultation not only with Aboriginal communities but also with ecologists knowledgeable in the contam inated environment is essential for protecting these important habitats. Furthermore, NEBA should also consider sociological factors because of public concerns over matters such as the return of a high amenity sites (e.g. highly-valued beaches and watercourses) to their pre-spill state.

Research is needed to further understanding of the effects of spilled oil in Arctic and permafrost ecosystems and how best to develop appropriate response strategies that mitigate the damage without further harm. Little has been done to advance knowledge of how to deal with a subsurface blowout in ice-covered marine environments. While the N ational Energy Board has identified the need for additional precautionary measures (e.g., same season relief well capacity), further research is needed to aid in development of oil detection and response decision support . With the advent of Arctic drilling for oil and hydrocarbons, improved methods are needed to detect and monitor the behaviour and transport of spills on, in and under ice and within the water column in the event of a subsurface deep sea blowout. In addition, oil interactions with permafrost and spring melt are associated with a number of challenges and unknowns that include site access for spill response, rapid spreading of oil during freshet, slow rates of weathering and how hydrocarbons interact with ice and suspended particulate matter.

Current technologies, equipment and response personnel are needed to be fully capable of effectively responding to major inland or offshore spills in northern Canada and Arctic regions. To advance operational spill response guidelines in these extreme envir onments, improvements in the understanding of the ecological effects of various oil spill cleanup measures need to be developed.

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More research is needed to rapidly determine the highest -risk combination of environmental characteristics when dealing with di lbit. For example, further studies are needed to understand under what circumstances a dilbit product is most likely to sink and its weathering would be slowest. In addition, more research is needed to understand the behaviour of dilbit and other heavy fuel oils in a range of environments, including near -shore and offshore marine, estuarine, and freshwater lakes, rivers and wetlands , under various combinations of climate, water chemistry and biological community conditions. This new information will hel p to optimize the selection of current spill response strategies and to support the development and validation of emerging technologies, including biodegradation of mousse (water-in-oil emulsions). Details on the factors that influence mouss es and tar balls are needed to better understand their formation, properties and treatment.

Based on the Panel's review of the physical and chemical properties of petroleum hydrocarbons, the overall behaviour of both conventional and unconventional oils currently transpo rted within Canada fall along a continuum for which existing oil spill technologies have been used. Unfortunately, a full understanding of the fate and transport of conventional and unconventional oil, such as dilbit, and associated impacts under different environmental conditions is insufficient. This compromises our ability of select and develop suitable response strategies and cleanup equipment.

In terms of priorities, just as conventional oils have been studied for years, greater focus should be given to dilbit, heavy fuel oils, emulsified oils and possibly tar balls in a range of environments, with special emphasis on identified high -risk coastal areas and cold northern parts of Canada. The results of these studies will help to identify and predict the level of impact following a spill, aid in the selection of the optimal spill response strategies, support the development and validation of emerging technologies and provide needed assistance in the assessment and understanding of the risks following a high impact spill.

Most literature on oil persistence is from marine environments. More information on oil persistence in riverine, lacustrine and mud flat environments would be beneficial for risk assessments and response operations. Also, more information i s needed on high gradient, high energy rivers and hyporheic zones connected to groundwater resources where there is a potential for mixing of oil into water by physical dispersion and widespread distribution.

Emphasis has recently been given to *in situ* 'green' technologies for the remediation of oil - contaminated sites that have higher public acceptance. Although much progress has been made over the last decade, green spill response technologies , such as phytoremediation (a technique based on enhanced oil biodegradation rates by plants and their associated rhizosphere microorganisms) and biodispersion (by potentially eco-friendly dispersants based on biosurfactants which are surface-active chemicals produced by microorganisms) should be tested and validated as cost -effective cleanup options. Although anaerobic biodegradation of hydrocarbons is still poorly understood, methods to accelerate the process are being considered for the treatment of oil sequestered within riverine, lacustrine, estuarine or marine sediments where it could persist for decades. Innovative methods (such as introduction of bioventing, air sparging or dissolved air flotation as response options) to stimulate anaerobic processes in the environment are needed to accelerate the rate of oil -impacted sediment remediation and habitat recovery.

Responses to oil spills usually consist of a series of dynamic, time -sensitive and complex processes subject to time and resource limitations, as well as environmental and technical constraints. The effectiveness of the response relies on t he efficient use of available information (oil, weather, wave, etc.) and resources (devices, manpower, money, etc.) in response decision -making and implementation. Inadequate response decision support is one of the major challenges that limit the efficiency of current response practices. Due to limited attention and investment, e xisting response decision support systems are rare and lack dynamic and interactive support from other assessment tools (e.g. risk analysis, impact assessment, spill modeling, process simulation, etc.) and field validation. These are especially true for the Arctic regions and waters where harsh environmental conditions limit the time period available for response.

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Question 6. Given the current state of the science, what are the priorities for research investments?

The behaviour, fate and effects of spilled oil are highly site-specific and decisions on oil spill cleanup and on assessing environmental and socio economic impacts are highly dependent on science. As illustrated in this RSC Panel report, the science is often lacking or highly uncertain due to the large number of variables that affect oil behaviour, fate and effects. There is an urgent need in Canada to develop science-based guidance and protocols for oil spill impact, risk assessments and cleanup. Guidance documents should provide scientific information and technical data on the various types of hydrocarbons transported, their potential effects and baseline information on potential sites exposed to high-risk -- material essential for the application of science-based decisions in oil spill response operations. This initiative requires a national level of coordinated scientific research in both the laboratory and the field (including experimental field trials with controlled releases of oil) that involves both support and funding from Indigenous communities, federal and provincial government agencies, private sector industries, academia and the public. The core needs of this proposed program are listed within seven high-priority research themes identified by the RSC Panel to improve spill prevention and the capability and effectiveness of oil spill response operations in Canada:

1. *Research to better understand the environmental impact of spilled crude oil in high-risk and poorly understood areas, such as Arctic waters, the deep ocean and shores or inland rivers and wetlands.*

2. *Research to increase the understanding of effects of oil spills on aquatic organisms at the population, community and ecosystem levels.*

3. *A national, priority-directed program of baseline research and monitoring to develop an understanding of the environmental and ecological characteristics of areas that may be affected by oil spills in the future and to identify any unique sensitivity to oil effects.*

4. *A program of controlled field research to better understand spill behaviour and effects across a spectrum of crude oil types in different ecosystems and conditions.*

5. *Research to investigate the efficacy of spill responses and to take full advantage of 'spills of opportunity'.*

6. *Research to improve spill prevention and develop/apply response decision support systems to ensure sound response decisions and effectiveness.*

7. *Research to update and refine risk assessment protocols for oil spills in Canada.*

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APPENDIX A. ROUTINE LABORATORY ANALYTICAL METHODS FOR CHEMICAL CHARACTERIZATION OF CRUDE OIL CONTAMINATION IN THE ENVIRONMENT

Crude oils are complex mixtures of thousands of chemicals; various combinations and proportions of these chemicals impart different properties on the oils (Chapter 2). Analysis of such mixtures, particularly complex high molecular weight (HMW) components, including resins and asphaltenes, remains technically challenging. No single method is sufficient to reveal and identify all components simultaneously and for some purposes it is unnecessary. Therefore, methods including those described below can be used singly or in combination to provide a range of information from bulk composition to detailed chemical 'fingerprints' of a crude oil spilled in the environment. A summary of analytical methods used in spill response (a subset of all available methods) is pre sented below and in **Table A1**. A recent review of additional methods used for analysis of crude oils ('petroleomics') is available (Rogers and McKenna 2011).

The concept of Total Petroleum Hydrocarbons (TPH) is important in oil analysis. TPH is a single value for each distinct crude oil, and represents the total mass of all hydrocarbons in that oil, including the volatile and extractable (non-volatile) hydrocarbons that are recovered and detected using a specific method. When used in the context of an enviro nmental sample, TPH often refers to the total hydrocarbons that can be extracted from the sample (EPH or extractable petroleum hydrocarbons). Many methods are available, and each will yield a different TPH value for a sample depending on the properties tha t the method analyzes, as described below. Therefore, the TPH value must be accompanied by a definition of the method used to obtain the value, and only recognized, standard methods, such as those vetted and archived by ASTM International (www.astm.org; formerly the American Society for Testing and Materials), the US-EPA, and ISO standards, should be used.

A potential complication relevant to spilled oils recovered from the environment for analysis is that non-petroleum hydrocarbons, such as plant oils and waxes, may be co -extracted from soils rich in vegetation or plant debris, and subsequently be detected by methods outlined below, thus contributing to TPH values without accurately reflecting the oil itself.

Infrared (IR) spectroscopy is a simple, non -destructive, rapid and inexpensive analytical technique developed for analysis of oils and greases that provides bulk composition data and allows estimation of physical properties of oils (Khanmohammadi et al. 2012). Typically, petroleum is extracted from an e nvironmental sample (usually water or soil) using a solvent (historically Freon 113, now replaced with a non -fluorocarbon solvent like cyclohexane; ASTM [2011]), and a beam of infrared light is passed through the extract. Absorption of particular IR wavelengths yields a spectrum based on the type and abundance of chemical bonds in the extract and provides a measure of TPH. In theory, most hydrocarbons will be detected quantitatively using this analysis, except for the very light and very heavy compounds bec ause the former may evaporate and the latter may not be soluble in cyclohexane. A drawback is that the standard method as applied to crude oils does not identify specific chemicals in a mixture and the presence of water (e.g., in oil samples recovered from the environment) interfe res with accurate measurements.

Fluorescence spectroscopy (FS) is simple, non -destructive and non -specific, like IR, but employs ultraviolet (UV) wavelengths (Steffens et al. 2011). When some aromatic hydrocarbons are irradiated with UV they re-emit that light at a different wavelength. This light can be captured and analyzed as a proxy for TPH concentration. The simplicity of the measurement lends itself to remote sensing of crude oil in aquatic systems. Several instruments in diff erent configurations have been developed to detect and estimate petroleum concentrations in real time. For heavy oils the fluorescence signal is broad and short -lived, although it is more robust for lighter oils (Steffens et al. 2011). FS only gives an est imate of the aromatics present in a sample and thus is inaccurate for estimating total hydrocarbons.

Gravimetric analysis is a rapid, inexpensive laboratory technique for determining TPH, typically after solvent extraction of the oil from an environmental sample. The mass of material that dissolves in the solvent and then remains after the solvent is evaporated is weighed. Unless combined with another analytical method, gravimetry cannot determine which specific chemicals within a class are present (or have been lost to the environment), but it can determine bulk changes in the oil and therefore indicate changes in gross properties. This method is unsuitable for light oils or dilbits that suffer large evaporative losses, but is more appropriate for heavier crudes and may be particularly useful for analyzing oil samples that present difficulties for other techniques, such as heavy oils and bitumens that have high resin and asphaltene contents and resist detailed chemical characterization. However, the measurement will include any non-petroleum materials in the sample that are extracted by the solvent used.

Liquid-Solid Column Chromatography and Fractionation. Oil that is recovered from the environment using extraction solvents subsequently may be subjected to chromatographic fractionation. Typically, the light fractions in the oil are depleted by artificial weathering in a laboratory fume hood ('topping' the oil) to make subsequent steps easier and because these light molecules would be lost during the fractionation procedure anyway. After determining the mass of oil that has been lost during topping (by mass difference), the asphaltene fraction is removed by precipitation using either n-pentane or n-hexanes, dried and weighed. The de-asphalted oil (also called the 'maltenes') is then passed through a glass column containing adsorbent(s) like silica. Discrete fractions (e.g., SARA; Chapter 2) comprising chemicals in different solubility classes are then eluted using solvents of different polarity (**Figure A1**). The masses of the fractions are determined after evaporating the solvents so that gravimetric changes, such as those caused by weathering (Chapter 2), can be discerned. Additionally, the saturate and aromatic fractions can be subjected to detailed chemical analysis using, for example, GC-FID or GC-MS, described below, to yield additional insight into the oil's composition. Thus, chromatography and fractionation can provide quantitative information, as well as composition, within technical limitations of detection.

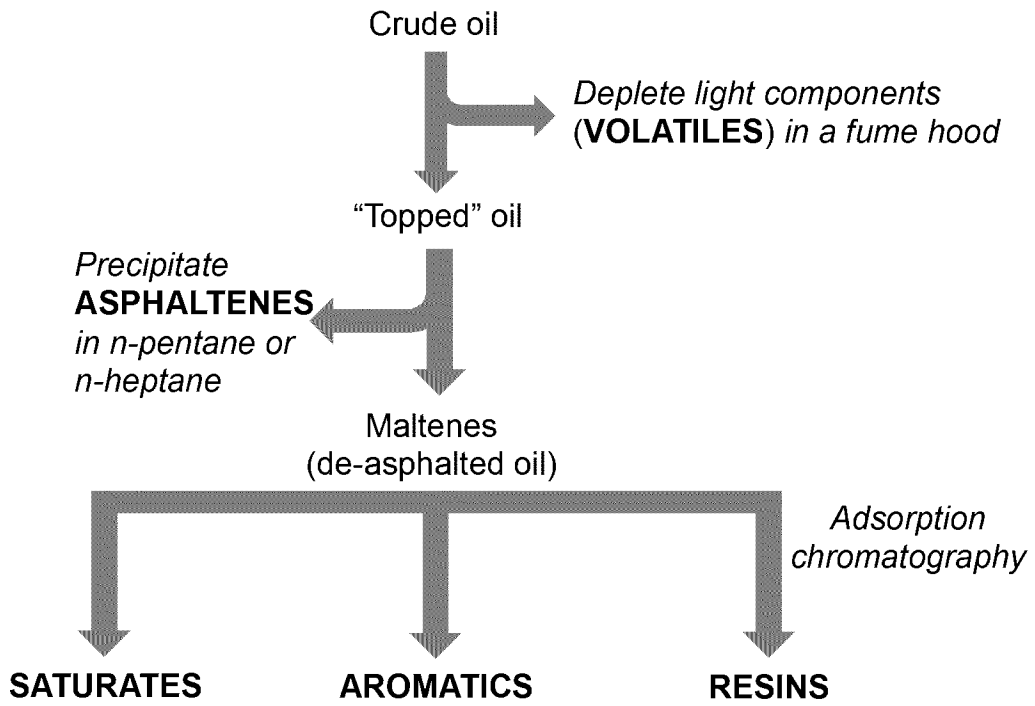

Figure A1 Liquid-solid chromatography and fractionation yielding SARA (saturates, aromatics, resins and asphaltenes) fractions of crude oil (Chapter 2), plus volatile hydrocarbons determined from mass of initial oil minus mass of topped oil.

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Gas Chromatography (GC) uses principles similar to those of the adsorption chromatography fractionation method described above, but uses elevated temperature and gas flow across the adsorbent phase rather than liquid solvents at ambient temperature to separate individual chemicals in the oil. Long, small-bore capillary columns (e.g., fused silica columns of 30 m length, 0.25 mm diameter with an ultra-thin adsorbent film of 0.25 μm) enable separation of complex chemical mixtures. Different adsorbents are available to separate specific chemical classes, providing a characteristic 'fingerprint' (chromatogram) of the oil where, ideally, each peak represents a single chemical (**Figure A2**). In practice, a finite range of molecular sizes and composition having boiling points below 524 °C can be detected using GC, routinely from C_6 to C_{30}. Only materials having no or low polarity are resolved, i.e., many saturates and aromatics, but few resins (asphaltenes are removed before GC analysis to prevent fouling of the column and instrument). Therefore, TPH values obtained by this method are limited to only GC-detectable and -resolvable compounds. This is important when analyzing weathering losses from heavy oils in which the large majority of material is not GC-detectable, because any changes observed by GC represent only minor mass changes in the total oil composition.

GC methods are in common use because of the broad range of hydrocarbons that are detected selectively and sensitively, and because identification of individual compounds can be achieved by coupling various detectors to GC analysis, including flame ionization detectors (GC-FID) and mass spectrometers (GC-MS), among others.

GC-FID uses a non-selective hydrogen flame to ionize and detect a broad range of compounds that are separated by and elute from the GC column. It is a sensitive and reproducible method commonly used for detecting ~C_6 to C_{30} hydrocarbons and some resins, but is not suitable for large, polar molecules such as asphaltenes. It is commonly applied to analysis of the most readily biodegradable fractions, the saturates and aromatics. However, even within these fractions, the presence of myriad isomers can render resolution of individual compounds difficult; in that case, the chromatogram is characterized by a 'hump' comprising unresolved peaks called the Unresolved Complex Mixture (UCM). The UCM is the dominant feature in chromatograms of heavy oils. Examples of GC-FID chromatograms are shown in **Figure A2**.

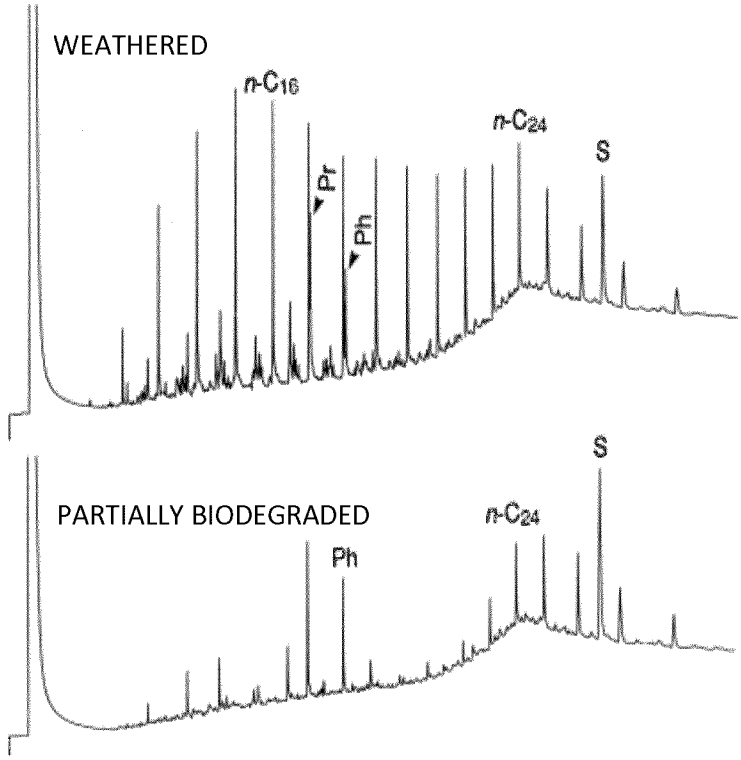

Figure A2. Examples of GC -FID chromatograms s howing the GC -detectable saturate fraction of an Alaska North Slope crude oil. *Top: weathered oil, showing evaporative loss of 'light ends' below C 12; Bottom: weathered and partially biodegraded oil, showing selective removal of n -alkanes below C 24. Numbered peaks: n -C_{16} and n -C_{24}= n -hexadecane and n -tetracosane, respectively; Pr and Ph= pristane and phytane, respectively (biomarkers that are slow to degrade); S= squalane, a reference hydrocarbon added to the oil as an internal standard. The remaining talle r peaks represent the n -alkane series, differing by a single –CH_2 group, and the shorter peaks represent multiple different isomers of iso - and cyclo - alkanes. The 'hump' toward the right -hand side represents the unresolved complex mixture (UCM), comprising isomers that are not resolved by the GC conditions used.*

GC-MS: The mass -selective detector yields highly detailed information about individual chemicals as they elute from the GC column, including inferred compound identity. However, although it is very sensitive, it may not be able to discern differences between multiple isomers that are common in crude oils. GC-MS is not used to measure bulk properties, such as TPH, but is more of a 'fine tool' to discern specific GC-detectable changes in oil composition.

A major disadvantage of all GC -based methods is that only certain classes of chemicals are resolved on a given adsorbent column and the detectors are only suitable for certain classes of petroleum components. For example, whereas saturated and aromatic hydrocarbons within a molecular weight range (e.g., ~C_6–C_{30}) are routinely detected using GC -FID and GC -MS, larger saturated and aromatic compounds and the resins and asphaltene fractions are particularly difficult to resolve and/or are undetectable by eit her method. GC -MS is particularly useful for routine analysis of changes in the specific chemical composition of an oil as it weathers or biodegrades, but usually does not provide comprehensive information, as it does not detect or identify large compounds.

An important consideration of GC analysis is that 'disappearance' of a peak from the oil chromatogram ('oil fingerprint') does not necessarily mean that the corresponding chemical has been completely removed from the environment. It is possible that the compound has been only slightly altered, e.g., by photooxidation (see Chapter 2) or by partial biodegradation by microbes, with the majority of the hydrocarbon skeleton remaining intact but being too polar to be detected

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or resolved by GC. Thus, GC chromat ograms must be analyzed carefully so as not to over - interpret the fingerprint and thereby overestimate degradation.

High Temperature Simulated Distillation by Gas Chromatography (GC -HT SimDis or HTSD) separates the components of petroleum in the order of t heir boiling points (Roussis and Fitzgerald 2000), generating bulk compositional information about the oil. It is particularly useful for analysis of oils with high contents of waxes, resins and asphaltenes that cannot be measured using conventional GC met hods. The curve generated by detecting boiling ranges as the analytical temperature increases to >700 °C can be used to compare different oils or to refer to calibration curves to estimate the oil's composition. The shape of the boiling point distribution curve (i.e., cumulative % mass loss with increasing temperature) reflects the proportions of low - and high-boiling point components in the oil.

An example is shown in **Figure A3** , comparing HTSD for a conventional oil (ASMB), an intermediate fuel oil (IFO-180) and two dilbit samples (Cold Lake and AWB **; Table 2.2**). A simplistic interpretation of the distillation curves is that a greater proportion of ASMB components have lower boiling points (are lower molecular weight) than those of IFO -180; Cold Lake and AWB have similar compositions and include light ends present in much lower proportions than the bulk of the oil and are similar in content to those in ASMB.

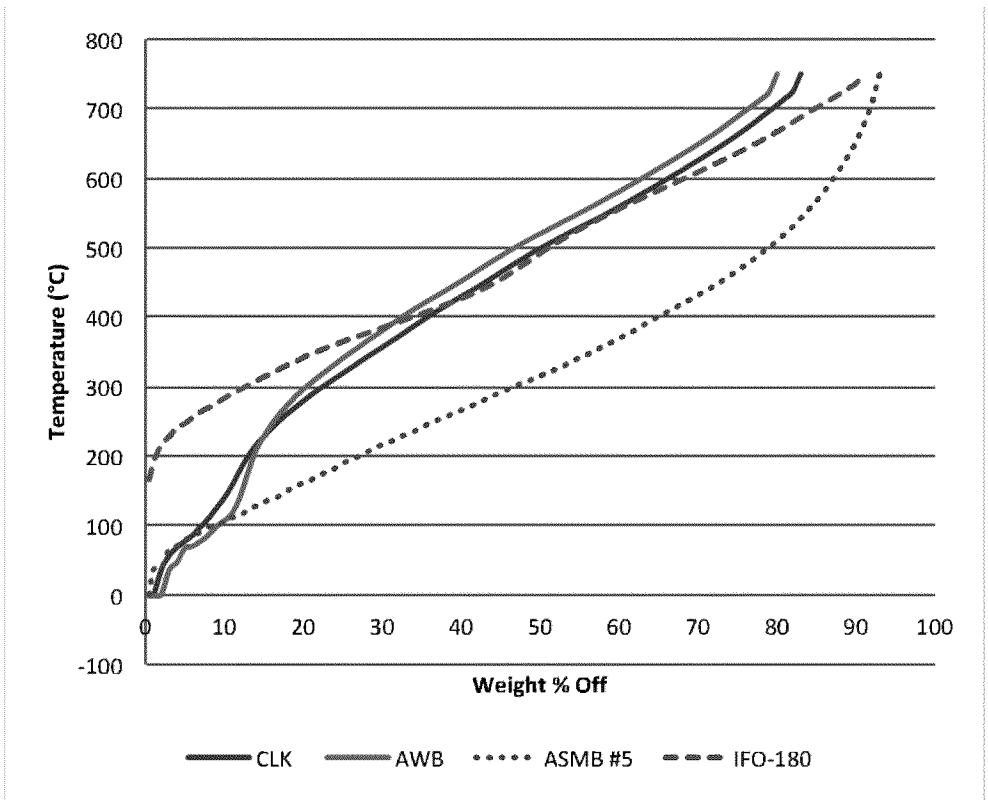

Figure A3. High Temperature Simulated Distillation *(HTSD) plot showing the simulated boiling point distributions for Cold Lake (CLK) and Access Western Blend (AWB) dilbits, Alberta Sweet Mixed Blend (ASMB) conventional crude and Intermediate Fuel Oil 180; see Table 2.2 for oil descriptions. Figure courtesy of Dr. H. Dettman, National Resources Canada – CanmetENERGY, Devon, Alberta.*

Non-routine analytical techniques

Two other techniques used for petroleum analysis are ElectroSpray Ionization Fourier transform ion cyclotron resonance mass spectrometry (ESI FT-ICR-MS) and Nuclear Magnetic Resonance (NMR) (Rogers and McKenna 2011). Both are highly sophisticated and require extremely

expensive instruments, as well as skilled operators, but they provide fundamental insights into petroleum composition that is not afforded by more routine analytical techniques.

Other methods are available (e.g., high performance liquid chromatography [HPLC]; Table A2) but they are typically used for more specialized analysis.

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Table A1. Summary of chemical analysis methods commonly used in oil spills

Name of method	Description of method	Advantage of the method	Disadvantage of the method	Reference
Infrared spectroscopy (IR)	• Provides miscellaneous information about complex mixtures (e.g., hydroxyl and carbonyl groups)	• Can analyze the hydrogen bonding in the crude oil mixture • Measurement of a great number of structural parameters • Helps determine oil age	• Gives more qualitative than quantitative information • Cannot detect some compounds • In some cases should be combined with other spectroscopic techniques, such as NMR spectroscopy • Includes standards only for 'oils and greases', not specifically for petroleum	Brown et al. 1975; Butt et al. 1986; Fernández-Varela et al. 2006; Gautam et al. 1998; Lynch and Brown 1973
Gas Chromatography (GC)	• Separates and analyzes complex oil components that can be vaporized without decomposition • Sample is carried through a column by the gas phase; chemical and physical properties of components differentiate their rate passing through the column, resulting in different retention times • Separated substances leaving the column are quantitatively detected by an electronic signal, which can be later detected by using detectors with different sensitivity and selectivity, such as flame photometric detector (FPD), flame ionization detector (FID), and t he thermal conductivity detector (TCD).	• Quick analysis; small sampling amount • Well analyze light oil • FID: very sensitive and selective • TCD: universal detector; low selectivity, low sensitivity • FPD: identify heavily biodegraded oils, sulphur heterocycles and others	• Limited use in heavy crude oil analysis • FID: ideal for limited compounds; compounds are destroyed • TCD: not very sensitive; compounds are destroyed	Butt et al. 1986; de Andrade et al. 2010; Gaines et al. 1999; Kenkel 2002; Kim et al. 2013; Payne et al. 2008; Simanzhenkov and Idem 2003; Yim et al. 2011
Mass spectrometry (MS)	• Gives information on compound structures, such as molecular weight and formula • Used for compound identification and new compound detection • MS techniques include electron impac t (EI), chemical ionization (CI), field ionization (FI), and fast atom bombardment (FAB) • Combine with chromatographic methods: GC - MS, HPLC-MS	• Identify compound structure	• Instrument can be expensive • Results may require technically sophisticated interpretation	Eide and Zahlsen 2005; Li et al. 1997; Rostad 2006; Sun et al. 2009; Tzing et al. 2003; Wang et al. 2002
High performance liquid chromatography (HPLC)	• Use high pressure to force chemical compounds to pass through a metal column • Detector used: UV absorption (UV), refractive index (RI) or fluorescence (F)	• Can analyze compounds with low volatility that are inappropriate for GC analysis • Can analyze some compounds that GC is unable to analyze (e.g., heavy crude oil) with precision in a short time	• Limited use in crude oil analysis due to the limited compound species determined • Unable to detect some compounds, depending on the detector used • UV and F only detect some species; F lacks sensitivity a nd is influenced by temperature	Fish et al. 1984; Kenkel 2002; Pasadakis et al. 2001; Speight and Speight 2002

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Table B1 Major properties of oil, analytical methods and relevance to oil behaviour

Parameter	Description of parameter	Analysis method	Relevance with application and cleanup measurements	References
Viscosity	• Measure of flow resistance • Liquids with lower viscosity flow faster • Dependent on oil composition; hence, the viscosities of crude oils vary. • Oils with greater saturates and aromatics fractions and lower asphaltenes and resins content have lower viscosity • Weathering of oil increases its viscosity • Generally exponentially dependent on temperature	• Viscometer: determines the flow rate of the oil under standard conditions; higher flow rate indicates lower viscosity • Rheometer: measures liquid flow in response to applied force • ASTM method D445	• Viscous oil spreads on water surface slowly and penetrates soils/sediments slowly • Oils with high viscosity affect the efficacy of pumps and skimmers	Aske et al. 2002; Beal 1946; Beggs and Robinson 1975; Lambert 2003; Morrison and Murphy 2006; ASTM 2015
Density	• Mass of a unit volume of oil, expressed as g/mL or kg/m^3 • Density of oil typically 0.7-0.99 g/mL, (versus seawater density; 1.03 g/mL) • Most oils float on seawater, except for a few bitumens • Heavily weathered oils tend to have greater density • Oil density is more temperature-dependent than is water	• Density meter (densitometer) • ASTM method D 5002	• Light grade and heavy oils • Submerged oil detection (e.g., acoustic remote sensing); surface oil detection (e.g., optical remote sensing) • Heavier oils are more persistent and may pose greater challenge for remediation	Aske et al. 2002; ASTM 2005; Fingas et al. 2006;
Specific gravity (relative density)	• Ratio of oil density to receiving water density • Varies with temperature • Quality indicator of oil by American Petroleum Institute (API), or API gravity	• ASTM method D287 -12b: Standard test method for API gravity of crude petroleum and petroleum products (hydrometer method)	• Gives approximate hydrocarbon composition and heat of combustion • Affects oil storage, handing and combustion	ASTM 2012a; Fingas 2010
Surface tension	• Net stresses at the boundaries between different substances • Expressed as the increased energy per unit area, or as force per unit length (mN/m)	• Tensiometer • du Noüy ring method • ASTM method D971	• Affects oil spreading • Relates to the final size of the oil slick • Lower surface tension leads to thinner and larger extent of slick	Aske et al. 2002; ASTM 2012b; du Noüy 1925;
Flash point	• The temperature at which the vapor over the liquid can be ignited • Flammable liquid	• Flash point analyzer • ASTM method D1310 (for low viscosity oil), and ASTM method D93 (for heavier oil)	• Safety indicator for spill cleanup process (light fuels such as gasoline can ignite under most ambient conditions)	ASTM 2007; ASTM 2013
Pour point	• The temperature at which oil ceases to flow in a five second-period • Ranges from -60 °C to 30 °C for crude oils • Lighter, less viscous oils have lower pour points	• ASTM method D97	• Limited use as an indicator of oil status • Important for oil recovery and transport	ASTM 2012b

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APPENDIX C

Table C1: Additional details of composition and properties of relevant crude oils and refined products. *Data excerpted from various sources**

Oil type (source province or country)	API gravity (°)	specific gravity at 15°C	% sulphur	% saturates	% resins	% asphalt-enes	% waxes	adhesion (g/m^2)*	% dispersion (agent)
Condensates and refined products used as diluents									
Sable Island condensate (NS)	40	0.82	0.03	88	0	1			
Fort Saskatchewan condensate blend (AB)	68	0.67	0.14	73					
Cold Lake Diluent (AB)	69	0.70	0.25						
Naphtha (AB)	57	0.75	0.07	85					
Southern Lights Condensate Diluent (US)	80	0.68	0.03						
Suncor Synthetic Crude Crude Oil/ SCO (AB)	33	0.86	0.2	82	1	0			40 (Corexit 9500)
Light crude oils									
Alberta Sweet Mixed Blend (ASMB) Reference oil #4 (AB)	36	0.84	0.6	65	5	33	4	13-61	40 (Corexit 9500)
Macondo (MC252) (USA)	35	0.84	0.3				0		
Norman Wells (NWT)	38	0.83	0.4	86	2	1	2		35 (Corexit 9500)
Transmountain Blend (BC)	34	0.86	0.8	81	2	4	7		
Statfjord (Norway)	38	0.84	0.3	68	6	2	8	14-62	40 (Corexit 9500)
Arabian Light (Saudi Arabia)	32	0.86	1.9	76	6	4	3	17-35	19 (Corexit 9500)
West Texas Intermediate (USA)	36-41	0.85	0..9	79	6	1	3	12-32	28 (Corexit 9500)

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Medium crude oils									
Alaska North Slope (ANS) middle pipeline (USA)	30	0.89	1.1	52	9	5		28-33	46% (Corexit 9500)
Atkinson Point (NWT)	24	0.91	0.9	48	14	3	1		
Prudhoe Bay-A (USA)	29	0.88	1	53	10	4	4	24-29	10 (Corexit 9500)
Gullfaks, Troll (Norway)	29	0.88	0.3	60	5	1	4	23-35	25 (Corexit 9500)
Shale oils									
Bakken (USA)	42		variable; usually <0.3						
Waxy crude oils									
Hibernia (NL)	35	0.85		73	7	2	5	12-160	11-21 (Corexit 9500)
Terra Nova (NL)	36	0.85	0.4	62	6	2	9	10-80	14
Sour crude oils									
Midale (SK)	31		1.5	51	9	5	5		25 (Corexit 9500)
BC Light (BC)	41	0.82	0.6						
Western Canadian Blend (WCB) (AB)	22	0.92	3						
West Texas Sour (USA)	30	0.78	1.5	51	9	5	5	21-25	25 (Corexit 9500)
Heavy oils and diluted bitumen									
Bunker C fuel oil (Fuel Oil No. 6) (international)	11	0.99	0.5	25	17	11	2	85-421	6-14 (Corexit 9500)
Cold Lake blend (dilbit) (AB)	23	0.92	4.7					13-977	
Albian Heavy Synthetic (AB)	20	0.94	2.3	42	26	6			15 (Corexit 9527)
Access Western Blend (AWB) (AB)	23	0.92	4						

Wabasca Heavy (AB)	16	0.96	5	41	28	9			<10 (Corexit 9527)
Western Canadian Select (WCS) (AB)	19-22	0.93	3						
Synbit Blend (AB)	21	0.93	2.9						
Athabasca bitumen (undiluted) (AB)	8	1.01	5			5			
Cold Lake bitumen (undiluted) (AB)	9.8	1.0002	6.9			13	2		0 (Corexit 9527)
Orimulsion-100 ultraheavy oil and water (Venezuela)	8	1.01	2.3	17	16	20			

**** range of adhesion values represents different weathered states***

REFERENCES

Chapter 1

AER (Alberta Energy Regulator). 2013. Pipeline Performance in Alberta 1990-2012. Report 2013-B. August, 2013. https://www.aer.ca/documents/reports/R2013-B.pdf.

AER. 2014a. Pace Oil and Gas Ltd. Wellhead Piping Failure Licence No. W0057420. May 19, 2012. Alberta Energy Regulator Investigation Report. https://www.aer.ca/documents/reports/IR_20140303-PaceOilandGas.pdf

AER. 2014b. Plains Midstream Canada ULC NPS 12 Rangeland South. Pipeline Failures and Release into the Red Deer River. Licence No. 5844, Line 1. June 7, 2012. Alberta Energy Regulator Investigation Report. https://www.aer.ca/documents/reports/IR_20140304-PlainsRangeland.pdf.

Anderson, C.M., Mayes, M. and R. Labelle. 2012. Update of Occurrence Rates for Offshore Oil Spills. USDI BOEM and BSEE, Herndon, VA.

API (American Petroleum Institute). 2009. Analysis of U.S. Spillage. API Publication No. 356. American Petroleum Institute, Washington, D.C.

Boudreau, P.R., Gordon, D.C. Jr., Harding, G.C., Loder, J.W., Black, J., Bowen, W.D., Campana, S., Cranford, P.J., Drinkwater, K.F., Van Eeckhaute, L., Gavaris, S., Hannah, C.G., Harrison, G., Hunt, J.J. McMillan, J., Melvin, G.D., Milligan, T.G., Muschenheim, D.K., Neilson, J.D., Page, F.H., Pezzack, D.S., Robert, G., Sameoto, D. and H. Stone. 1999. The Possible Environmental Impacts of Petroleum Exploration Activities on the Georges Bank Ecosystem. Canadian Technical Report of Fisheries and Aquatic Sciences 2259.

CBC. 2013. Tanker truck crashes into BC creek. *CBC News* July 27, 2013. http://www.cbc.ca/news/canada/british-columbia/jet-fuel-spill-evacuation-order-lifted-in-b-c-1.1306998.

Christopherson, S. and K. Dave. 2014. A New Era of Crude Oil Transport: Risks and Impacts in the Great Lakes Basin, Community and Regional Development Institute Reports. Issue Number 15. Department of Development Sociology. Cornell University.

ERCB (Energy Resources Conservation Board). 2011. Penn West Petroleum Ltd. Well Blowout 14-20-064-10W5M. Energy Resources Conservation Board Investigation Report. https://www.aer.ca/documents/reports/IR_20110823_PennWest.pdf

ERCB. 2012. Midway Energy Ltd. Hydraulic Fracturing Incident: Interwellbore Communication. January 13, 2012. Energy Resources Conservation Board Investigation Report. https://www.aer.ca/documents/reports/IR_20121212_Midway.pdf.

ERCB. 2013. Plains Midstream Canada ULC NPS 20 Rainbow Pipeline Failure. License No. 5592, Line No. 1. April 28, 2011. Energy Resources Conservation Board Investigation Report. Calgary, AB. https://www.aer.ca/documents/reports/IR_20130226-PlainsMidstream.pdf.

EPA. 2015. Galena Train Derailment. U.S. Environmental Protection Agency. EPA in Illinois. http://www2.epa.gov/il/galena-train-derailment.

Executive Flight Centre. 2014. Lemon Creek Environmental Monitoring Plan Update. August 14, 2014. http://www.lemoncreekresponse.ca/index.php.

Frittelli, J., Andrews, A., Parkomak, P., Pirog, R., Ramseur, J/L. and M. Ratner. 2014. U.S. Rail Transportation of Crude Oil: Background and Issues for Congress. Congressional Research Service 7-5700. R43390.

Gillis, L. 2015. Water advisory finally lifted in Gogama derailment area. *Timmins Press* June 10, 2015. http://www.timminspress.com/2015/06/10/water-advisory-finally-lifted-in-gogama-derailment-area.

Government of Saskatchewan. 2015. Petroleum and Natural Gas Spill Report Directory. http://www.qp.gov.sk.ca/Spills/Spill%20Report%20Directory.pdf

Hurley, G.V. and J. Ellis. 2004. Environmental Effects of Exploratory Drilling Offshore Canada: EEM Data and Literature Review. Final Report Prepared for the Canadian Environmental Assessment Agency Regulatory Advisory Committee (RAC).

Independent Panel Review. 2014. Independent Panel Review of the Canadian Natural Primrose Flow to Surface Causation Report. http://www.aer.ca/documents/reports/CNRLPrimrose_PanelReport_201407.pdf

ITOPF. 2014. Oil Tanker Spill Statistics 2014. International Tanker Owners Pollution Federation. http://www.itopf.com/knowledge-resources/documents-guides/document/oil-tanker-spill-statistics-2013/.

ITOPF. 2015a. Atlantic Empress Case Study. International Tanker Owners Pollution Federation. http://www.itopf.com/in-action/case-studies/case-study/atlantic-empress-west-indies-1979/).

ITOPF. 2015b. Amoco-Cadiz Case Study. International Tanker Owners Pollution Federation. http://www.itopf.com/in-action/case-studies/case-study/amoco-cadiz-france-1978/).

ITOPF. 2015c. Odyssey Case Study. International Tanker Owners Pollution Federation. http://www.itopf.com/in-action/case-studies/case-study/odyssey-off-canada-1988/).

ITOPF. 2015d. Tanker incident statistics. International Tanker Owners Pollution Federation. http://www.itopf.com/knowledge-resources/data-statistics/statistics/

Lee, K., Prince, R.C., Greer, C.W., Doe, K.G., Wilson, J.E.H., Cobanli, S.E., Wohlgeschaffen, G.D., Alroumi, D., King. T. and G.H. Tremblay. 2003. Composition and toxicity of residual Bunker C fuel oil in intertidal sediments after 30 years. *Spill Science and Technology Bulletin* 8: 187-199.

LeFever, J. 2008. What's happening at Parshall, North Dakota. *North Dakota Department of Mineral Resources Newsletter* 35 (1): 1-2.

Mackenzie, D. 2011. Operation clean-up. *New Scientist* 212 (2836): 46-49.

Mangione, K. 2015. CN train derails south of Timmins, Ont. *CTV News* February 15, 2015. http://www.ctvnews.ca/canada/cn-train-derails-south-of-timmins-ont-1.2237139.

Martin, J.D., Adam, J., Hollebone, B., King, T., Brown, R.S. and P.V. Hodson. 2014. Chronic toxicity of heavy fuel oils to fish embryos using multiple exposure scenarios. *Environmental Toxicology and Chemistry* 33: 677-687.

National Energy Board. 2015. Canadian Crude Oil Exports by Rail. https://www.neb-one.gc.ca/nrg/sttstc/crdlndptrlmprdct/stt/cndncrdlxprtsrl-eng.html.

NRC (National Research Council). 2003. Oil in the Sea III: Inputs, Fates and Effects. National Academies Press, Washington, DC. http://www.nap.edu/openbook.php?isbn=0309084385.

NTSB (US National Transportation Safety Board). 2013. Accident Involving Two Freight Trains Casselton, North Dakota. United States National Transportation Safety Board Preliminary Report. http://www.ntsb.gov/investigations/AccidentReports/Reports/Casselton_ND_Preliminary.pdf

NTSB. 2014. National Transportation Safety Board Safety Recommendation. January 23, 2014. http://www.ntsb.gov/safety/safety-recs/RecLetters/R-14-001-003.pdf.

NTSB. 2015. National Transportation Safety Board Docket. NTSB Accident ID DSA14MR004, Dec 30, 2013, Casselton, ND. Public release date April 27, 2015. http://dms.ntsb.gov/pubdms/search/hitlist.cfm?docketID=55926&CurrentPage=1&EndRow=15&StartRow=1&order=1&sort=0&TXTSEARCHT=) .

Owens, E.H., Taylor, E., Marty, R. and D.I. Little. 1993. An inland oil spill response manual to minimize adverse environmental impacts. In: *Proceedings of 1993 International Oil Spill Conference*. American Petroleum Institute, Washington, DC. Pp.105-109.

Owens, E.H., Prince, R.C. and R.B. Taylor. 2008. Natural attenuation of heavy oil on a coarse sediment beach: results from Black Duck Cove, Nova Scotia, Canada over 35 years following the Arrow oil spill. In: *Proceedings of the Thirty-first AMOP Technical Seminar on Environmental Contamination and Response, June 3-5, 2008, Calgary, Alberta.* pp. 585-599.

Raby, J. 2015. CSX provides update on W.Va. oil train derailment cleanup. *Washington Post* July 21, 2015. http://www.washingtontimes.com/news/2015/jul/21/csx-to-discuss-west-virginia-oil-train-derailment-/#!

Riordan Seville, L., Federico-O'Murchuand, S. And T. Connor. 2015. Heimdal, North Dakota, evacuated after fiery oil train crash. *NBC News* May 6, 2015. http://www.nbcnews.com/news/us-news/heimdal-north-dakota-evacuated-after-fiery-oil-train-crash-n354686

SL Ross Environmental Research Ltd. 2014. Guideline Document for Measuring the Biological Effects of Accidental Oil Spills. Report to Fisheries and Oceans Canada, National Contaminants Advisory Group, Ottawa, ON.

Spies, R.B., Rice, S.D., Wolfe, D.A. and B.A. Wright. 1996. The effect of the Exxon Valdez oil spill on Alaskan coastal environment. In: *Proceedings of the 1993 Exxon Valdez Oil Spill Symposium*. American Fisheries Society, Bethesda, MD. Pp. 1-16.

ED_001324_00007647-00391

Thormann, M.N. and S.E. Bayley. 2008. Impacts of the CN Rail Oil Spill on Softstem Bulrush-Dominated Lacustrine Marshes in Wabamun Lake. Report submitted to Alberta Environment, Edmonton, AB.

TSB (Transportation Safety Board of Canada). 2001. Crude Oil Pipeline Rupture. Enbridge Pipelines Inc. 864-Millimetre Line ¾, Mile Post 109.42 Near Hardisty, Alberta. 17 January 2001. Transportation Safety Board of Canada Pipeline Investigation Report P01H0004. http://www.tsb.gc.ca/eng/rapports-reports/pipeline/2001/p01h0004/p01h0004.asp.

TSB. 2013a. Runaway and Main-Track Derailment. Montreal, Maine and Atlantic Railway Freight Train MMA-002 Mile 0.23, Sherbrooke Subdivision. Lac Megantic, Quebec. 06 July 2013. Transportation Safety Board of Canada Railway Investigation Report R13D0054. http://www.tsb.gc.ca/eng/rapports-reports/rail/2013/r13d0054/r13d0054.pdf.

TSB. 2013b. Statistical Summary Pipeline Occurrences 2013. Transportation Safety Board of Canada. http://www.bst-tsb.gc.ca/eng/stats/pipeline/2013/sspo-2013.pdf.

TSB. 2015a. Statistical Summary Railway Occurrences 2014. Transportation Safety Board of Canada. http://www.tsb.gc.ca/eng/stats/rail/2014/sser-ssro-2014.pdf

TSB 2015b Derailment and Fire of Second Canadian National Crude Oil Train Near Gogama, Ontario. http://www.tsb.gc.ca/eng/medias-media/communiques/rail/2015/r15h0021-20150317.asp

TSPS (Tanker Safety Panel Secretariat). 2013. **A Review of Canada's Ship-Source Oil Spill Preparedness and Response Regime: Setting the Course for the Future. Ottawa.** https://www.tc.gc.ca/media/documents/mosprr/transport_canada_tanker_report_accessible_eng.pdf

Van Pelt, K. 2015. Update 5 – CSX train hauling North Dakota oil derails, cars ablaze in W. Virginia. *Reuters* February 17, 2015. http://uk.reuters.com/article/2015/02/17/usa-train-derailment-csx-idUKL1N0VQ0P820150217.

Wilhelm, S.I., Robertson, G.J., Ryan, P.C. and D.C. Schneider. 2007. Comparing an estimate of seabirds at risk to a mortality estimate from the November 2004 Terra Nova FPSO oil spill. *Marine Pollution Bulletin* 54: 537-544.

WSP. 2014. Risk Assessment for Marine Spills in Canadian Water: Phase 1, Oil Spills South of the 60th Parallel. Report to Transport Canada. http://files.wspdigital.com/risk/oil/english/131-17593-00_ERA_Oil-Spill-South_150116.pdf.

Young, L. 2014. Crude oil spills are bigger from trains than pipelines. *Global News* January 8, 2014. http://globalnews.ca/news/1069624/how-do-crude-spills-compare-by-rail-truck-pipeline-you-may-be-surprised/

Chapter 2

Abbasnezhad, H., Foght, J.M. and M.R. Gray. 2011a. Adhesion to the hydrocarbon phase increases phenanthrene degradation by *Pseudomonas fluorescens* LP6a. *Biodegradation* 22: 485-496.

Abbasnezhad, H., Gray, M. and J.M. Foght. 2011b. Influence of adhesion on aerobic biodegradation and bioremediation of liquid hydrocarbons. *Applied Microbiology and Biotechnology* 92: 653-675.

Advanced Technology, Inc. and Continental Shelf Associates, Inc. 1990. The Physical Persistence of Spilled Oil: An Analysis of Oil Spills Previous to *Exxon Valdez*. National Oceanic and Atmospheric Administration (NOAA). http://www.oil-spill-info.com/Publications/1990_Persistence_Spilled_Oil_NOAA_rpt.pdf (accessed June 2015)

Allen, E. W. 2008. Process water treatment in Canada's oil sands industry: I. Target pollutants and treatment objectives. *Journal of Environmental Engineering and Science* 7: 123–138.

AMAP (Arctic Monitoring and Assessment Programme). 1998. Chapter 10: Petroleum Hydrocarbons. AMAP Assessment Report: Arctic Pollution Issues. Oslo, Norway. http://www.amap.no/documents/doc/amap-assessment-report-arctic-pollution-issues/68

American Fuel & Petrochemical Manufacturers. 2014. A Survey of Bakken Crude Oil Characteristics Assembled for the U.S. Department of Transportation. http://www.ourenergypolicy.org/wp-content/uploads/2015/06/Survey-of-Crude-Oil-Characteristics_FINAL-1.pdf (accessed June, 2015)

Andersen, S.I. and J.G. Speight. 2001. Petroleum resins: Separation, character and role in petroleum. *Petroleum Science and Technology* 19:1-34

AOSRT (Arctic Oil Spill Response Technology – Joint Industry Programme). 2014. Environmental Impacts of Arctic Oil Spills and Arctic Oil Spill Response Technologies. Literature Review and Recommendations. Arctic Oil Spill Response Joint Industry Programme. http://www.arcticresponsetechnology.org/research-projects.

Atlas, R.M. and T.C. Hazen. 2011. Oil biodegradation and bioremediation: A tale of the two worst spills in U.S. History. *Environmental Science & Technology* 45: 6709-6715.

Azam, F. and F. Malfatti. 2007. Microbial structuring of marine ecosystems. *Nature Reviews Microbiology* 5:782-792.

Bertoncini, F., Courtiade Tholance, M. and D. Thiebaut. 2013. Gas Chromatography and 2D-Gas Chromatography for Petroleum Industry: The Race for Selectivity. IFP Energies Nouvelles Publications. Editions Technip, Paris.

Blenkinsopp, S., Wang, A., Foght, J., Westlake, D.W.S., Sergy, G., Fingas, M., Landriault, M., Sigouin, L. and K. Semple. 1996. Assessment of the Freshwater Biodegradation Potential of Oils Commonly Transported in Alaska. Environment Canada Report. ASPS# 95:0065.

Bobra, M. and E.J. Tennyson. 1989. Photooxidation of petroleum. In: Boehm, P.D. and J.W. Farrington (eds.) *Proceedings of the 12th Arctic and Marine Oilspill Program (AMOP) Technical Seminar.* Environmental Protection Service, Ottawa, Canada, pp. 129-147.

Boek, E.S., Headen, T.F. and J.T. Padding. 2010. Multi-scale simulation of asphaltene aggregation and deposition in capillary flow. *Faraday Discussions* 144: 271-284

Boufadel, M.C., Sharifi, Y., Van Aken, B., Wrenn, B.A. and K. Lee. 2010. Nutrient and oxygen concentrations within the sediments of an Alaskan beach polluted with the *Exxon Valdez* oil spill. *Environmental Science & Technology* 44: 7418-7424.

BP (British Petroleum). 2013. Gulf Science Data MC-252 Oil Characterization Data File. Reference No. O-01v01-01. http://gulfsciencedata.bp.com/go/doctype/6145/178706 (accessed June 2015)

Brakstad, O.G., Nordtug, T. and M. Throne-Holst. 2015. Biodegradation of dispersed Macondo oil in seawater at low temperature and different oil droplet sizes. *Marine Pollution Bulletin* 93 (1-2): 144-152.

Brandvik, P.J., and L.-G. Faksness. 2009. Weathering processes in Arctic oil spills: Meso-scale experiments with different ice conditions. *Cold Regions Science and Technology* 55: 160-166.

Brooks, G.R., Larson, R.A., Schwing, P.T., Romero, I., Moore, C., Reichart, G.-J., Jilbert, T., Chanton, J.P., Hastings, D.W., Overholt, W.A., Marks, K.P. Kostka, J.E., Holmes, C.W. and D. Hollander. 2015. Sedimentation pulse in the NE Gulf of Mexico following the 2010 DWH blowout. *PLOS One* 10(7): e0132341.

Buskey, N. 2013. 40,000-pound tar mat unearthed on Grand Terre. Houma Today, Houma, LA. http://www. houmatoday.com/article/20130703/ARTICLES/130709834. (accessed Sept, 2015)

Caumette, G., Lienemann, C.-P., Merdrignac, I., Bouyssiere, B., and R. Lobinski. 2009. Element speciation analysis of petroleum and related materials. *Journal of Analytical Atomic Spectrometry* 24: 263-276.

Chanton, J., Zhao, T., Rosenheim, B.E., Joye, S., Bosman, S., Brunner, C., Yeager, K.M., Diercks, A.R. and D. Hollander. 2014. Using natural abundance radiocarbon to trace the flux of petrocarbon to the seafloor following the Deepwater Horizon oil spill. *Environmental Science & Technology* 49: 847-854.

Chronopoulou, P.M., Sanni, G.O., Silas-Olu, D.I., van der Meer, J.R., Timmis, K.N., Brussard, C.P.D. and T.J. McGenity. 2015. Generalist hydrocarbon-degrading bacterial communities in the oil-polluted water column of the North Sea. *Microbial Biotechnology* 8: 434-447.

Clemente, J.S. and P.M. Fedorak. 2005. A review of the occurrence, analyses, toxicity, and biodegradation of naphthenic acids. *Chemosphere* 60: 585–600.

Crosby, S., Fay, R., Groark, C., Kani, A., Smith, J. R., Sullivan, T. and R. Pavia. 2013. Transporting Alberta Oil Sands Products: Defining the Issues and Assessing the Risks. U.S. Dept. of Commerce, NOAA Technical Memorandum NOS OR&R 43. Emergency Response Division, NOAA, Seattle, WA.

http://permanent.access.gpo.gov/gpo45878/noaa_oil_sands_report_09.2013.pdf
Crude Quality Inc. 2015. crudemonitor.ca. http://crudemonitor.ca. (accessed June 2015)
Daling, P.S., Moldestad, M.Ø., Johansen, Ø., Lewis, A. and R. Rødal. 2003. Norwegian testing of emulsion properties at sea – the importance of oil type and release conditions. *Spill Science & Technology Bulletin* 8(2): 123-136.
Davenport, C. 2015. U.S. will allow drilling for oil in Arctic Ocean. *New York Times*, May 11, 2015. http://www.nytimes.com/2015/05/12/us/white-house-gives-conditional-approval-for-shell-to-drill-in-arctic.html?_r=0 (Accessed August 2015)
de Araujo, P.L.B., Mansoori, G.A. and E.S. de Araujo. 2012. Diamondoids: occurrence in fossil fuels, applications in petroleum exploration and fouling in petroleum production. A review paper. *International Journal of Oil, Gas and Coal Technology* 5(4): 316–367.
Deppe, U., Richnow, H.H. Michaelis, W. and G. Antranikian. 2005. Degradation of crude oil by an arctic microbial consortium. *Extremophiles* 9:461-470.
Dettman, H., Inman, A. and S. Salmon. 2005. Chemical characterization of GPC fractions of Athabasca bitumen asphaltenes isolated before and after thermal treatment. *Energy & Fuels* 19: 1399-1404.
Dettman, H. and G. Irvine. 2015. Comparison of oil-in-water emulsion stability of diluted bitumen, light and heavy crude oils. Abstract, *38th Arctic and Marine Oilspill Program (AMOP) Technical Seminar.* Environment Canada, Ottawa.
Dew, W.A., Hontela, A., Rood, S.B. and G.G. Pyle. 2015. Biological effects and toxicity of diluted bitumen and its constituents in freshwater systems. *Journal of Applied Toxicology. (in press)* doi: 10.1002/jat.3196.
Dickins, D. 2011. Behavior of oil spills in ice and implications for Arctic spill response. Paper OTC 22126. Arctic Technology Conference Houston, Texas, USA, 7–9 February 2011. http://www.arcticresponsetechnology.org/wp-content/uploads/2012/11/Dickins-Review-Paper.pdf. (accessed June, 2015)
Dorobantu, L.S., Yeung, A.K.C., Foght, J.M. and M.R. Gray. 2004.Stabilization of oil-water emulsions by hydrophobic bacteria. *Applied and Environmental Microbiology* 70(10): 6333-6336.
Ehrenhauser, F.S., Avij, P., Shu, X., Dugas, V., Woodson, I., Liyana-Arachchi, T., Zhang, A., Hung, F.R. and K.T. Valsaraj. 2014. Bubble bursting as an aerosol generation mechanism during an oil spill in the deep-sea environment: laboratory experimental demonstration of the transport pathway. *Environmental Science: Processes and Impacts* 16: 65-73
Environmental Technology Centre, Environment Canada. 2015. Oil Properties Database. http://www.etc-cte.ec.gc.ca/databases/oilproperties/ (accessed June 2015)
Fingas, M.F. 2004. Modeling evaporation using models that are not boundary-layer regulated. *Journal of Hazardous Materials* 107: 27-36.
Fingas, M.F. 2014-2015. Bitumens and diluted bitumens From Western Canadian Oil Sands: Chapters 1-9. *International Spill Control Organization (ISCO) Newsletters* 457-465. http://www.spillcontrol.org/ (accessed May, 2015)
Fingas, M.F. 2014. Water-in-oil emulsions: Formation and prediction. *Journal of Petroleum Science Research* 3(1): 38-49.
Fingas, M.F. 2015a. Appendix C. Ice nomenclature. In: Fingas, M.F. (ed.) *Handbook of Oil Spill Science and Technology*. Wiley & Sons, Inc., Hoboken, NJ.
Fingas, M.F. 2015b. Chapter 1. Introduction to oil chemistry and properties. In: Fingas, M.F. (ed.) *Handbook of Oil Spill Science and Technology*. Wiley & Sons, Inc., Hoboken, NJ.
Fingas, M.F. 2015c. Chapter 7: Oil and petroleum evaporation. In: Fingas, M.F. (ed.) *Handbook of Oil Spill Science and Technology*. Wiley & Sons, Inc., Hoboken, NJ.
Fingas, M.F. 2015d. Diluted bitumen (Dilbit): A future high risk spilled material. *Interspill 2015 Conference Proceedings, Amsterdam, March 24-26, 2015.*
Fingas, M.F. 2015e. Review of the properties and behaviour of diluted bitumens. In: *Proceedings of the 38th Arctic and Marine Oilspill Program (AMOP) Technical Seminar*. Environment Canada, Ottawa. pp. 470-494.
Fingas, M.F. and B. Fieldhouse. 2015. Chapter 8: Water-in-oil emulsions: Formation and prediction. In:

ED_001324_00007647-00394

Fingas, M.F.(ed.) *Handbook of Oil Spill Science and Technology*. Wiley & Sons, Inc., Hoboken, NJ.

FISG (Federal Interagency Solutions Group, Oil Budget Calculator Science and Engineering Team). 2010. Oil Budget Calculator Deepwater Horizon. Technical Documentation. A report to the National Incident Command. http://www.restorethegulf.gov/sites/default/files/documents/pdf/OilBudgetCalc_Full_HQ-Print_111110.pdf. (accessed August 2015)

Fitzpatrick, F.A., Boufadel, M.C., Johnson, R., Lee, K., Graan, T.P., Bejarano, A.C., Zhu, Z., Waterman, D., Capone, D.M., Hayter, E., Hamilton, S.K., Dekker, T., Garcia, M.H. and J.S. Hassan. 2015. Oil-Particle Interactions and Submergence from Crude Oil Spills in Marine and Freshwater Environments – Review of the Science and Future Science Needs. U.S. Geological Survey Open File Report 2015-1076. http://dx.doi.org/10.3133/ofr20151076.

Foght, J. 2008. Anaerobic biodegradation of aromatic hydrocarbons: Pathways and prospects. *Journal of Molecular Microbiology and Biotechnology* 15: 93-120.

Fowler, S.W., Readman, J.W., Oregioni, B., Villeneuve, J.-P. and K. McKay. 1993. Petroleum hydrocarbons and trace metals in nearshore Gulf sediments and biota before and after the 1991 war: An assessment of temporal and spatial trends. *Marine Pollution Bulletin* 27: 171-182.

Fustic, M., Bennett, B., Huang, H. and S. Larter. 2012. Differential entrapment of charged oil – New insights on McMurray Formation oil trapping mechanisms. *Marine and Petroleum Geology* 36: 50-69.

González-Macías, C., Schifter, I., Lluch-Cota, D.B., Méndez-Rodríguez, L., and S. Hernández-Vázquez. 2006. Distribution, enrichment and accumulation of heavy metals in coastal sediments of Salina Cruz Bay, Mexico. *Environmental* Monitoring *and Assessment* 118: 211-230.

Gosselin, P., Hrudey, S.E., Naeth, M.A., Plourde, A., Therrien, R., Van Der Kraak, G. and Z. Xu. 2010. Environmental and Health Impacts of Canada's Oil Sands Industry. Royal Society of Canada Expert Panel Report. http://www.rsc.ca/en/expert-panels/rsc-reports. (Accessed May 2015).

Government of Canada. 2013. Properties, Composition and Marine Spill Behaviour, Fate and Transport of Two Diluted Bitumen Products from the Canadian Oil Sands. Federal Government Technical Report En84-96/2013E-PDF ISBN 978-1-100-230004-7. http://publications.gc.ca/site/eng/457066/publication.html.

Gray, M.R. 2015. Upgrading Oilsands Bitumen and Heavy Oil. University of Alberta Press, Edmonton, AB.

Greer, C.W., Fortin, N., Sanschagrin, S., Yergeau, E., Whyte, L.G., King, T.L. and K. Lee. 2015. Natural degradation potential for crude oil at sub-zero temperatures in the Canadian Arctic marine environment. Abstract, *38th Arctic and Marine Oilspill Program (AMOP) Technical Seminar.* Environment Canada, Ottawa.

Grewer, D.M., Young, R.F., Whittal, R.M. and P.M. Fedorak. 2010. Naphthenic acids and other acid-extractables in water samples from Alberta: What is being measured? *Science of the Total Environment* 408: 5997–6010.

Gros, J., Nabi, D., Würz, B., Wick, L.Y., Brussaard, C.P.D., Huisman, J., van der Meer, J.R., Reddy, C.M. and J.S. Arey. 2014. First day of an oil spill on the open sea: Early mass transfers of hydrocarbons to air and water. *Environmental Science & Technology* 48: 9400-9411.

Harper, J.R., Laforest, S. and G. Sergey. 2015. Field investigations of intertidal sediment permeability related to spilled oil retention in British Columbia shorelines. In: *Proceedings of the 38th Arctic and Marine Oilspill Program (AMOP) Technical Seminar.* Environment Canada, Ottawa. Pp. 631-648.

Hazen, T.C., Dubinsky, E.A., DeSantis, T.Z., Andersen, G.L., Piceno, Y.M., Singh, N., Jansson, J.K., Probst, A., Borglin, S.E., Fortney, J.L., Stringfellow, W.T., Bill, M., Conrad, M.E., Tom, L.M., Chavarria, K.L., Alusi, T.R., Lamendella, R., Joyner, D.C., Spier, C., Baelum, J., Auer, M., Zemla, M.L., Chakraborty, R., Sonnenthal, E.L., D'Haeseleer, P., Holman, H.Y.N., Osman, S., Lu, Z.M., Van Nostrand., J., Deng, Y., Zhou, J.Z. and O.U. Mason. 2010. Deep-sea oil plume enriches indigenous oil-degrading bacteria. *Science* 330: 204–208.

Head, I.M., Jones, D.M. and W.F.M. Röling. 2006. Marine microorganisms make a meal of oil. *Nature*

ED_001324_00007647-00395

Reviews Microbiology 4: 1730.
Hess Corporation. 2012. Safety Data Sheet No. 6608. http://www.hess.com/docs/us-safety-data-sheets/crude-oil-(sour).pdf?sfvrsn=2. (accessed June, 2015).
Hoff, A. and H.D. Dettman. 2012. Global asphaltene content and structure related to process behaviour. Paper WHOC12-275. In: *Proceedings, World Heavy Oil Congress*. Aberdeen, Scotland.
Hollebone, B.P. 2015. Appendix A. The oil properties data appendix. In: M.F. Fingas, M.F. (ed.) Handbook of Oil Spill Science and Technology. Wiley & Sons, Inc., Hoboken, NJ.
Hollebone, B.P., Fieldhouse, B., Sergey, G., Lamter, P., Wang, Z., Yang, C. and M. Landriaut. 2011. The behaviour of heavy oil in fresh water lakes. In: *Proceedings of the 34th Arctic and Marine Oilspill Program (AMOP) Technical Seminar on Environmental Contamination and Response.* Environment Canada, Ottawa. Pp. 668-709.
Indian and Northern Affairs Canada. 1995. Petroleum Exploration in Canada. Chapter 4 – Mackenzie Delta and Beauford Sea. http://www.aadnc-aandc.gc.ca/eng/1321378288752/1321378450630 (accessed August, 2015)
Islam, A., Kim, D., Yim, U.H., Shim, W.J. and S. Kim. 2015. Structure-dependent degradation of polar compounds in weathered oils observed by atmospheric pressure photo-ionization hydrogen/deuterium exchange ultrahigh resolution mass spectrometry. *Journal of Hazardous Materials* 296: 93-100.
ITOPF (International Tanker Owners Pollution Federation Ltd.,). 2014. Fate of Oil Spills: Weathering. http://www.itopf.com/knowledge-resources/documents-guides/fate-of-oil-spills/weathering/ (accessed July, 2015)
Jones, D., West, C.E., Scarlett, A.G., Frank, R.A. and S.J. Rowland. 2012. Isolation and estimation of the 'aromatic' naphthenic acid content of an oil sands process-affected water extract. *Journal of Chromatography A* 1247: 171-175.
Johnson, R.J., Smith, B.E., Sutton, P.A., McGenity, T.J., Rowland, S.J. and C. Whitby. 2011. Microbial biodegradation of aromatic alkanoic naphthenic acids is affected by the degree of alkyl side chain branching. *ISME Journal* 5: 486–496.
Jokuty, P., Whiticar, S., Fingas, M., Meyer, E. and C. Knobel. 1995. Hydrocarbon groups and their relationships to oil properties and behaviour. In: *Proceedings of the 18th Arctic and Marine Oilspill Program (AMOP) Technical Seminar.* Environment Canada, Ottawa. Pp. 1-19.
Joung, DJ. and A.M. Shiller. 2013. Trace element distribution in the water column near the Deepwater Horizon well blowout. *Environmental Science & Technology* 47: 2161-2168.
Joye, S.B. 2015. Deepwater Horizon, 5 years on. *Science* 349: 592-593.
Kang, Z., Yeung, A., Foght, J.M. and M.R. Gray. 2008a. Hydrophobic bacteria at the hexadecane-water interface: examination of micrometer-scale interfacial properties. *Colloids and Surfaces B: Biointerfaces* 67: 59-66.
Kang, Z., Yeung, A., Foght, J.M. and M.R. Gray. 2008b. Mechanical properties of hexadecane-water interfaces with adsorbed hydrophobic bacteria. *Colloids and Surfaces B: Biointerfaces* 62: 273-279.
Katakey, R. and W. Zhu. 2015. Shell halts Alaska offshore exploration after failing to find enough oil. *Bloomberg Business*, September 27, 2015. http://www.bloomberg.com/news/articles/2015-09-28/shell-to-stop-exploring-offshore-alaska-on-regulations-costs (accessed October 2015)
Kaufman, M. 2015. US Follows Royal Dutch Shell plc. Backs away from Arctic drilling. http://royaldutchshellplc.com/2015/10/19/us-follows-royal-dutch-shell-plc-backs-away-from-arctic-drilling/ (accessed October 19, 2015)
King, T.L., Robinson, B., Boufadel, M. and K. Lee. 2014. Flume tank studies to elucidate the fate and behavior of diluted bitumen spilled at sea. *Marine Pollution Bulletin* 83: 32-37.
King, T., Robinson, B., Ryan, S. and Y. Lu. 2015a. Fate of Chinese and Canadian oil treated with chemical dispersants in a wave tank. In: *Proceedings of the 38th Arctic and Marine Oilspill Program (AMOP) Technical Seminar*. Environment Canada, Ottawa. pp. 798-811.
King, T.L., Robinson, B., McIntyre, C., Toole, P., Ryan, S., Saleh, F., Boufadel, M.C. and K. Lee. 2015b. Fate of surface spills of Cold Lake blend diluted bitumen treated with dispersant and mineral fines in a wave tank. *Environmental Engineering Science* 32: 250-261.

Kleindienst, S., M. Seidel, K. Ziervogel, S. Grim, K. Loftis, S. Harrison, S.Y. Malkin, M.J. Perkins, J. Field, M.L. Sogin, T. Dittmar, U. Passow, P.M. Medeiros and S.B. Joye. 2015. Chemical dispersants can suppress the activity of natural oil-degrading microorganisms. *Proceedings National Academy of Sciences (USA)* in press. doi:10.1073/pnas.1507380112

Kokal, S. 2002. Crude oil emulsions: A state-of-the-art review. SPE 77497. Society of Petroleum Engineers. www.onepetro.org/journal-paper/SPE-77497-PA

Kropp, K.G., Saftic, S., Andersson, J.T. and P.M. Fedorak. 1996. Transformations of six isomers of dimethylbenzothiophene by three *Pseudomonas* strains. *Biodegradation* 7: 203-221.

Laforest, S., Lambert, P. and M. Goldthorp. 2015. The Shoreline Studies Program: An update on shoreline surveys and initial studies on the penetration and retention of diluted bitumen into marine shoreline substrates. In: *Proceedings of the 38th Arctic and Marine Oilspill Program (AMOP) Technical Seminar.* Environment Canada, Ottawa. Pp. 649-663.

Lea-Smith, D.J., Biller, S.J., Davey, M.P., Cotton, C.A.R., Perez Sepulveda, B.M., Turchyn, A.B., Scanlan, D.J., Smith, A.G., Chisholm, S.W. and C.J. Howe. 2015. Contribution of cyanobacterial alkane production to the ocean hydrocarbon cycle. *Proceedings National Academy of Sciences (USA)* in press. doi: 10.1073/pnas.1507274112

Lee, K. 2002. Oil–particle interactions in aquatic environments: Influence on the transport, fate, effect and remediation of oil spills. *Spill Science & Technology Bulletin* 8(1): 3-8.

Lee, K. and E.M. Levy. 1989. The enhanced biodegradation of condensate and crude oil on beaches of Atlantic Canada. In: *Proceedings 1989 Oil Spill Conference, February 13 - 16, 1989, San Antonio, Texas.* American Petroleum Institute, Environmental Protection Agency, and United States Coast Guard. pp. 479-486.

Lee, K., M. Boudreau, J. Bugden, L. Burridge, S.E. Cobanli, S. Courtenay, S. Grenon, B. Hollebone, P. Kepkay, Z. Li, M. Lyons, H. Niu, T.L. King, S. MacDonald, E.C. McIntyre, B. Robinson, S.A. Ryan and G. Wohlgeschaffen. 2011a. State of Knowledge Review of Fate and Effect of Oil in the Arctic Marine Environment 2011. National Energy Board of Canada, Ottawa, ON. 267 pp. https://docs.neb-one.gc.ca/ll-eng/llisapi.dll/fetch/2000/90463/621169/700096/704342/A2A8R2_-_NEB_State_of_Knowledge_Review_of_Fate_and_Effect_of_Oil_DFO_COOGER.pdf?nodeid=704343&vernum=-2

Lee, K., Li, Z., Robinson, B., Kepkay, P.E., Blouin, M. and B. Doyon. 2011b. Field trials of *in-situ* oil spill countermeasures in ice-infested waters. In: *Proceedings of the International Oil Spill Conference (IOSC).* http://dx.doi.org/10.7901/2169-3358-2011-1-160 (accessed July, 2015)

Lee, K., Bugden, J., Cobanli, S., King, T., McIntyre, C., Robinson, B., Ryan, S. and G. Wohlgeschaffen. 2012. UV- epifluorescence microscopy analysis of sediments recovered from the Kalamazoo River. U.S. EPA Kalamazoo Administrative Record, document #1277. http://www.epa.gov/enbridgespill/ar/enbridge-AR-1277.pdf (accessed July, 2015)

Madison, B.N., Hodson, P.V. and V.S. Langlois. 2015. Diluted bitumen causes deformities and molecular responses indicative of oxidative stress in Japanese medaka embryos. *Aquatic Toxicology* 165:222-230.

Martin, J.D., Adams, A., Hollebone, B.P., King, T., Brown, R.S. and P.V. Hodson. 2014. Chronic toxicity of heavy fuel oils to fish embryos using multiple exposure scenarios. *Environmental Toxicology and Chemistry* 33:677-687.

Mbadinga, S.M., Wang, L.-Y., Zhou, L, Liu, J.-F., Gu, J.-D., and B.-Z. Mu. 2011. Microbial communities involved in anaerobic degradation of alkanes. *International Biodegradation and Biodeterioration* 65: 1-13.

McFarlin, K.M., Prince, R.C., Perkins, R. and M.B. Leigh. 2014. Biodegradation of dispersed oil in Arctic seawater at -1°C. *PLoS ONE* 9: e84297.

McGowan, F. 1990. The development of Orimulsion and Venezuelan oil strategy. *Energy Policy* 18(10): 913-926.

Melbye, A.G., Brakstad, O.G., Hokstad, J.N., Gregersen, I.K., Hansen, B.H., Booth, A.M., Rowland, S.J. and K.E. Tollefsen. 2009. Chemical and toxicological characterization of an unresolved complex mixture-rich biodegraded oil. *Environmental Toxicology and Chemistry* 28(9): 1815-1824.

Murphy, D.W., Li, C., d'Albignac, V., Morra, D. and J. Katz. 2015. Splash behavior and oily marine aerosol production by raindrops impacting oil slicks. *Journal of Fluid Mechanics*. 780: 536-577.

Nmegbu, G.C.J. 2014. Analysis of effects of temperature on petrophysical properties of petroleum reservoir rocks and fluids. *International Journal of Engineering Research and Development* 10(6): 53-61.

North Dakota Petroleum Council. No date. Bakken Crude Properties. http://www.ndoil.org/resources/bkn/. (accessed June 2015)

NRC (National Research Council). 1999. Spills of Nonfloating Oils: Risk and Response. National Academy of Science, National Research Council, Washington, D.C. http://www.nap.edu/openbook.php?isbn=0309065909. (accessed June 2015).

NRC. 2003. Oil in the Sea III: Inputs, Fates and Effects. National Academies Press. http://www.nap.edu/catalog/10388.html. (accessed June, 2015).

NRC. 2013. Effects of Diluted Bitumen on Crude Oil Transmission Pipelines. National Academy of Sciences, Washington, D.C. http://www.nap.edu/read/18381/ (accessed Oct 2015)

NRC. 2014. Responding to Oil Spills in the U.S. Arctic Marine Environment. National Academies Press, Washington, D.C. http://www.nap.edu/catalog/18625/responding-tooil-spills-in-the-us-arctic-marine-environment. (Accessed June, 2015)

NTSB (National Transportation Safety Board). 2012. Enbridge Incorporated Hazardous Liquid Pipeline Rupture and Release. Accident Report NTSB/PAR-12/01, PB2012-916501. http://www.ntsb.gov/investigations/AccidentReports/Reports/PAR1201.pdf. (Accessed July 2015).

Omotoso, O.E., Munoz, V.A. and R.J. Mikula. 2002. Mechanisms of crude oil–mineral interactions. *Spill Science & Technology Bulletin* 8(1): 45-54

Owens, E.H., Harper, J.R. Robson, W. and P.D. Boehm. 1987. Fate and persistence of crude oil stranded on a sheltered beach. *Arctic* 40 (Supp. 1): 109-123.

Passow, U. 2014. Formation of rapidly-sinking, oil-associated marine snow. *Deep-Sea Research II:* doi:10.1016/j.dsr2.2014.10.001 (in press)

Passow, U., Ziervogel, K., Asper, V. and A. Diercks. 2012. Marine snow formation in the aftermath of the Deepwater Horizon oil spill in the Gulf of Mexico. *Environmental Research Letters* 7: 035301.

Payne, J.R., McNabb Jr., G.D. and J.R. Clayton Jr. 1990. Oil weathering behavior in Arctic environments. *Polar Research* 10: 632-662.

Peng, P., Morales-Izquierdo, A., Hogg, A. and O.P. Strausz. 1997. Molecular structure of Athabasca asphaltene: Sulfide, ether and ester linkages. *Energy & Fuels* 11: 1171-1187.

Peters, K.E., Walters, C.C. and J.M. Moldowan. 2005. The Biomarker Guide. Vol. 2. Biomarkers and Isotopes in Petroleum Exploration and Earth History. 2nd Edition. Cambridge University Press.

Petrich, C., Karlsson, J. and H. Eicken. 2013. Porosity of growing sea ice and potential for oil entrainment. *Cold Regions Science and Technology* 87: 27-32.

Petrova, L.M. Abbakumova, N.A., Foss, T.R. and G.V. Romanov. 2011. Structural features of asphaltene and petroleum resin fractions. *Petroleum Chemistry* 51(4): 252-256.

Prince, R.C. 2015. Oil spill dispersants: Boon or bane? *Environmental Science & Technology* 49: 6376-6384.

Prince, R.C. and C.C. Walters. 2007. Chapter 11. Biodegradation of oil hydrocarbons and its implications for source identification. In: Wang, Z. and S.A. Stout (eds.) *Oil Spill Environmental Forensics: Fingerprinting and Source Identification*. Academic Press, Burlington MA. pp. 349-380.

Romero, I.C., Schwing, P.T., Brooks, G.R., Larson, R.A., Hastings, D.W., Ellis, G., Goddard, E.A. and D.J. Hollander. 2015. Hydrocarbons in deep-sea sediments following the 2010 Deepwater Horizon blowout in the northeast Gulf of Mexico. *PLOS ONE*, 10(5): e0128371

Rowland, S.J., Scarlett, A.G., Jones, D., West, C.E. and R.A. Frank. 2011. Diamonds in the rough: Identification of individual naphthenic acids in oil sands process water. *Environmental Science & Technology* 45: 3154-3159.

Santos-Echeandía, J., Prego, R. and A. Cobelo-García. 2008. Influence of the heavy fuel spill from the *Prestige* tanker wreckage in the overlying seawater column levels of copper, nickel and vanadium (NE Atlantic Ocean). *Journal of Marine Systems* 72: 350-357.

Segato, R. no date. Quality Guidelines for Western Canadian Condensate. Canadian Association of

ED_001324_00007647-00398

Petroleum Producers (CAPP). http://www.coqa-inc.org/docs/default-source/meeting-presentations/Segato0608.pdf. (accessed June, 2015)

Sergy, G.A. and P.J. Blackall. 1987. Design and conclusions of the Baffin Island Oil Spill project. *Arctic* 40 (supplement): 1-9. http://arctic.journalhosting.ucalgary.ca/arctic/index.php/arctic/index

Sheremata, J.M., Gray, M.R., Dettman, H.D. and W.C. McCaffrey. 2004. Quantitative molecular representation and sequential optimization of Athabasca asphaltenes. *Energy & Fuels* 18: 1377-1384.

Short, J.W. 2013. Susceptibility of Diluted Bitumen Products from the Alberta Tar Sands to Sinking in Water. Report to the National Energy Board, # D72-80-2. 21 pp. http://docs.neb-one.gc.ca/ll-eng/llisapi.dll?func=ll&objId=941166&objAction=browse (accessed July, 2015)

Sikkema, J., de Bont, J.A.M. and B. Poolman. 1995. Mechanisms of membrane toxicity of hydrocarbons. *Microbiology Reviews* 59: 201-222.

Simecek-Beatty, D. and W.J. Lehr. 2007. Chapter 13. Trajectory modeling of marine oil spills. In: Wang, Z. and S.A. Stout (eds.) *Oil Spill Environmental Forensics: Fingerprinting and Source Identification*. Academic Press, Burlington MA. pp. 405-418

Speight, J.G. 2004. Petroleum asphaltenes: Part 1, Asphaltenes, resins and the structure of petroleum. *Oil & Gas Science and Technology – Revue d'IFP* 59(5): 467-477.

Speight, J.G. 2014. The Chemistry and Technology of Petroleum. 5th Edition. CRC Press, Hoboken, NJ.

Stoffyn-Egli, P. and K. Lee. 2002. Formation and characterization of oil-mineral aggregates. *Spill Science & Technology Bulletin* 8(1): 31-44.

Strain, P. M. 1986. The persistence and mobility of a light crude oil in a sandy beach. *Marine Environmental Research* 19: 49-76.

Strausz, O.P., Mohelsky, T.W. and E.M. Lown. 1992. The molecular structure of asphaltene: an unfolding story. *Fuel* 71: 1355-1363.

Thibodeaux, L.J., Valsarag, K.T., John, V.T., Papadopoulos, K.D., Pratt, L.R. and N.S. Pesika. 2011. Marine oil fate: Knowledge gaps, basic research, and development needs; a perspective based on the Deepwater Horizon spill. *Environmental Engineering Science* 28(2): 87-93.

Tonina, D. and J.M. Buffington. 2007. Hyporheic exchange in gravel bed rivers with pool-riffle morphology: Laboratory experiments and three-dimensional modeling. *Water Resources Research* 43(W01421).

Tonina, D. and J.M. Buffington. 2009. Hyporheic exchange in mountain rivers I: Mechanics and environmental effects. *Geography Compass* 3(3): 1063-1086.

Tsaprailis, H. 2014. Properties of Dilbit and Conventional Crude Oils. Report to Alberta Innovates–Energy and Environmental Solutions. Alberta Innovates–Technology Futures. http://www.ai-ees.ca/media/10927/properties_of_dilbit_and_conventional_crude_oils_-_aitf_-_final_report_revised.pdf (accessed June, 2015).

US-EPA (US Environmental Protection Agency). 1994. Chemical summary for cyclohexane. http://www.epa.gov/chemfact/s_cycloh.txt (accessed June, 2015).

US-EPA. 2013a. Attachment 2: U.S. EPA Volume Estimate for Submerged Line 6B Oil in the Kalamazoo River. http://www.epa.gov/enbridgespill/pdfs/20130625/enbridge_epareview_20130508_qsoreport_attachment2.pdf (accessed July 4, 2015).

US-EPA. 2013b. Fact Sheet August 2013. Dredging begins on Kalamazoo River, Enbridge oil spill, Marshall, Michigan. http://epa.gov/enbridgespill/pdfs/enbridge_fs_201308.pdf (accessed August, 2015)

Valentine, D.L., Fisher, G.B., Bagby, S.C., Nelson, R.K., Reddy, C.M., Sylva, S.P. and M.A. Woo. 2014. Fallout plume of submerged oil from *Deepwater Horizon. Proceedings of the National Academy of Sciences (USA)* 111(45): 15906-15911.

Van Hamme, J.D., Singh, A. and O.P. Ward. 2003. Recent advances in petroleum microbiology. *Microbiology and Molecular Biology Reviews* 67: 503-549.

Venosa, A.D and E.L. Holder. 2007. Biodegradability of dispersed crude oil and two different temperatures. *Marine Pollution Bulletin* 54(5): 545-553.

Warnock, A.M., Hagen, S.C. and D.L. Passeri. 2015. Marine tar residues: a review. *Water, Air and Soil*

ED_001324_00007647-00399

Pollution 226: 68. DOI 10.1007/s11270-015-2298-5. (in press)

Wang, Z., Fingas, M., Blenkinsopp, S., Sergy, G., Landriault, M., Sigouin, L., Foght, J., Semple, K., and D.W.S. Westlake. 1998. Comparison of oil composition changes due to biodegradation and physical weathering in different oils. *Journal of Chromatography A* 809: 89-107.

Waterman, D.M. and M.H. García. 2015. Laboratory Tests of Oil-Particle Interactions in a Freshwater Riverine Environment with Cold Lake Blend Weathered Bitumen. Civil Engineering Studies; Hydraulic Engineering Series No. 106. Ven Te Chow Hydrosystems Laboratory, University of Illinois, Urbana IL.

Wei, A., Moldowan, J.M, Peters, K.E., Wang, Y. and W. Xiang. 2007. The abundance and distribution of diamondoids in biodegraded oils from the San Joaquin Valley: Implications for biodegradation of diamondoids in petroleum reservoirs. *Organic Geochemistry* 38: 1910-1926.

Welch, M.A. 2014. Northern energy hub? Pipeline to Churchill explored -- deepwater port ice-free longer. *Winnipeg Free Press*, August 1, 2014. http://www.winnipegfreepress.com/local/northern-energy-hub-239208921.html (accessed October 2015)

Whitby, C. 2010. Microbial naphthenic acid biodegradation. *Advances in Applied Microbiology* 70: 93-125.

Wikipedia. 2015. Western Canadian Select. http://en.wikipedia.org/wiki/Western_Canadian_Select. (accessed June 2015)

Winter, J. and R. Haddad. 2014. Ecological impacts of dilbit spills: Considerations for natural resource damage assessment. 37th Arctic and Marine Oil Spill Program (AMOP) Technical Seminar. Environment Canada, Ottawa. https://usresponserestoration.files.wordpress.com/2014/06/winter-and-haddad-dilbit-nrda-submitted-4-11-14.pdf.

Wise Jr., J.P., Wise, J.T.F., Wise, C.F., Wise, S.S., Gianios Jr., C., Xie, H., Thomson, W.D., Perkins, C., Falank, C. and J.P. Wise Sr. 2014. Concentrations of the genotoxic metals, chromium and nickel, in whales, tar balls, oil slicks and released oil from the Gulf of Mexico in the immediate aftermath of the Deepwater Horizon oil crisis: Is genotoxic metal exposure part of the Deepwater Horizon legacy? *Environmental Science & Technology* 48: 2997-3006.

WSP. 2014. Risk Assessment for Marine Spills in Canadian Water: Phase 1, Oil Spills South of the 60th Parallel. Report to Transport Canada. http://files.wspdigital.com/risk/oil/english/131-17593-00_ERA_Oil-Spill-South_150116.pdf.

Yang, C., Wang, Z., Yang, Z., Hollebone, B., Brown, C.E., Landriault, M., and B. Fieldhouse. 2011. Chemical fingerprints of Alberta oil sands and related petroleum products. *Environmental Forensics* 12: 173–188.

Yang, F., Tchoukov, P., Pensini, E., Dabros, T., Czarnecki, J., Masliyah, J. and Z. Xu. 2014. Asphaltene sub-fractions responsible for stabilizing water-in-oil emulsions: Part 1: Interfacial behaviours. *Energy & Fuels* 28(11): 6897-6904.

Yang, F., Tchoukov, P., Dettman, H., Tecklebrham, R.B., Liu, L, Dabros, T., Czarnecki, J. and Z. Xu. 2015. Asphaltene sub-fractions responsible for stabilizing water-in-oil emulsions: Part II: Molecular representations and molecular dynamic situations. *Energy & Fuels* doi: 10.1021/acs.energyfuels.5b00657 *(in press).*

Yarranton, H.W., Motahhari, H., Schoeggl, F.F. and Z.J. Zhou. 2015. Evaporative weathering of diluted bitumen films. 174557-PA SPE. *Journal of Canadian Petroleum Technology* 54: 233-244.

Yim, U.H., Kim, M., Ha, S.Y., Kim, S. and W.J. Shim. 2012. Oil spill environmental forensics: The *Heibei Spirit* oil spill case. *Environmental Science & Technology* 46: 6431-6437.

Zhang, X., Wiseman, S., Yu, H., Liu, H., Giesy, J.P. and M. Hecker. 2011. Assessing the toxicity of naphthenic acids using a microbial genome wide living cell reported array system. *Environmental Science & Technology* 45(5): 1984-1991.

Zhao, L., Boufadel, M.C., Socolofsky, S.A., Adams, E., King, T. and K. Lee. 2014. Evolution of droplets in subsea oil and gas blowouts: Development and validation of the numerical model VDROP. *Journal of Marine Pollution Bulletin* 83: 58-69.

Zhou, J., Dettman, H. and M. Bundred. 2015. A comparative analysis of environmental behavior of diluted bitumen and conventional crudes. In: *Proceedings of the 38th Arctic and Marine Oilspill Program*

ED_001324_00007647-00400

(AMOP) Technical Seminar. Environment Canada, Ottawa. pp. 495-516.

Chapter 3

Abed, R.M.M., Al-Thukair, A. and D. De Beer. 2006. Bacterial diversity of a cyanobacterial mat degrading petroleum compounds at elevated salinities and temperatures. *FEMS Microbiology Ecology* 57: 290–301.

Abu Laban, N., Dao, A., Semple, K. and J. Foght. 2014. Biodegradation of C_7 and C_8 isoalkanes under methanogenic conditions. *Environmental Microbiology:* (in press) doi:10.1111/1462-2920.12643.

Arrigo, K.R. 2005. Marine microorganisms and global nutrient cycles. *Nature* 437: 349-355.

Atlas, R.M. 1981. Microbial degradation of petroleum hydrocarbons: An environmental perspective.*Microbiology Reviews* 45: 180-209.

Atlas, R. M. (ed.). 1984. Petroleum Microbiology. Macmillan Co., New York.

Atlas, R. and T. Hazen. 2011. Oil biodegradation and bioremediation: A tale of the two worst spills in U.S. history.Environmental Science & Technology 45: 6709–6715.

Bartha, R. and I. Bossert. 1984. The treatment and disposal of petroleum wastes. In: Atlas, R.M. (ed.) Petroleum Microbiology. Macmillan, New York, NY. pp. 553–578.

Beal, R. and W.B. Betts. 2000. Role of rhamnolipid biosurfactants in the uptake and mineralization of hexadecane in *Pseudomonas aeruginosa*. *Journal of Applied Microbiology* 89: 158-168.

Belore, R. C. and I. Buist. 1988. Modelling of oil spills in snow. In: *Proceedings of the 11th Arctic Marine Oil Spill Program Technical Seminar, Vancouver, BC, 7-9 June*. Environment Canada, Ottawa, ON. Pp. 9-29.

Bosma, T.N.P., Middeldorp, P.J.M., Schraa, G. and A.J.B. Zender. 1997. Mass transfer limitation of biotransformation: quantifying bioavailability. *Environmental Science and Technology* 31: 248-252.

Bossert, I. and R. Bartha. 1984. The fate of petroleum in soil ecosystems. In: Atlas, R.M. (ed.) Petroleum Microbiology. Macmillan Publishing Company, New York. Pp. 435-476.

Boufadel, M.C., Suidan, M.T., Rauch, C.H.,Ahn, C.H. and A.D. Venosa. 1999. Nutrient transport in beaches subjected to freshwater input and tides. In: *Proceedings of 1999 International Oil Spill Conference*. American Petroleum Institute, Washington, DC. Pp. 471-476.

Bragg, J.R. and E.H. Owens. 1995. Shoreline cleansing by interactions between oil and fine mineral particles. In: *Proceedings of 1995 International Oil Spill Conference.* American Petroleum Institute, Washington, DC. Pp. 216-227.

Button, D. K. 1985. Kinetics of nutrient-limited transport and microbial-growth. *Microbiology Review* 49 (3): 270-297.

Caldwell, M.E., Garrett, R.M., Prince, R.C. and J.M. Suflita. 1998. Anaerobic biodegradation of long-chain n-alkanes under sulfate-reducing conditions. *Environmental Science and Technology* 32: 2191-2195.

Campo, P., Venosa, A.D. and M.T. Suidan. 2013. Biodegradability of Corexit 9500 and Dispersed South Louisiana Crude Oil at 5 and 25 °C. *Environmental Science and Technology* 47: 1960-1967.

Carls, M.G., Thomas, R.E., Lilly, M.R. and S.D. Rice. 2003. Mechanism for transport of oil-contaminated groundwater into pink salmon redds. *Marine Ecology Progress Series* 248: 245–255.

Cerniglia, C.E. 1992. Biodegradation of polycyclic aromatic hydrocarbons. *Biodegradation* 3: 351-368.

Chronopoulou, P.M., Gbemisola, O.S., Silas-Olu, D.I., van der Meer, J.R., Timmis, K.M., Brussaard, C.P.D. and T.J. McGenity. 2015. Generalist hydrocarbon-degrading bacterial communities in the oil-polluted water column of the North Sea. *Microbial Biotechnology* 8(3): 434-447.

Coates, J.D., Woodward, J., Allen, J., Philip, P. and D.R. Lovley. 1997. Anaerobic degradation of polycyclic hydrocarbons and alkanes in petroleum-contaminated marine harbour sediments. *Applied Environmental Microbiology* 63: 3589-3593.

ED_001324_00007647-00401

Colwell, R.R., Mills, A.L., Walker, J.D. Garcia-Tolle, P. and P.V. Campos. 1978. Microbial ecology studies of the Metula spill in the Straits of Magellan. Journal of the Fisheries Research Board of Canada 35: 573-580.

Cooney, J.J. 1984. The fate of petroleum pollutants in freshwater ecosystems. In: Atlas (ed) *Petroleum Microbiology*. Macmillan Publishing Company, New York. Pp. 355-398.

Das, N. and P. Chandran. 2011. Microbial Degradation of Petroleum Hydrocarbon Contaminants: An Overview. *Biotechnology Research International* 2011: 13 pp. http://dx.doi.org/10.4061/2011/941810.

Diaz, M.P., Boyd, K.G., Grigson, S.G.W. and J.G. Burgess. 2002. Biodegradation of crude oil across a wide range of salinities by an extremely halotolerant bacterial consortium MPD-M, immobilized onto polypropylene fibers. *Biotechnology and Bioengineering* 79(2):145–153.

Dickins, D.F. 2011.Behaviour of oil spills in ice and implications for spill response.In Proceedings of the Arctic Technology Conference, Houston TX, 7-9 February 2011, Offshore Technology Conference (OTC) Paper # 22126, pp.15.

Dutta, T. K., and S. Harayama. 2000. Fate of crude oil by the combination of photooxidation and biodegradation. Environ. Sci. Technol. 34:1500–1505.

Faksness, L.-G. and P. J. Brandvik. 2008a. Distribution of water soluble components from Arctic marine oil spills – A combined laboratory and field study. *Cold Regions Science and Technology* 54: 97-105.

Faksness, L.-G. and P. J. Brandvik. 2008b. Distribution of water soluble components from oil encapsulated in Arctic sea ice: Summary of three field seasons. *Cold Regions Science and Technology* 54: 106-114.

Fingas, M. 2015. Diluted bitumen (dilbit): a future high risk spilled material. In: Proceedings of the 2015 Interspill Conference, March 24-26, 2015, Amsterdam.

Floodgate, G. 1984. The fate of petroleum in marine ecosystems. In: Atlas, R.M. (ed.) *Petroleum Microbiology*. Macmillan Publishing Company, New York. Pp. 355-398.

Foght, J. 2006. Potential for biodegradation of sub-littoral residual oil by naturally occurring microorganisms following the Lake Wabamun train derailment. http://environment.gov.ab.ca/info/library/7746.pdf

Foght, J.M. 2008. Anaerobic biodegradation of aromatic hydrocarbons: Pathways and prospects. *Journal of Molecular Microbiology and Biotechnology* 15: 93-120.

Foght, J.M., Fedorak, P.M., and D.W.S. Westlake. 1990. Mineralization of ^{14}C-hexadecane and ^{14}C-phenanthrene in crude oil: specificity among bacterial isolates. *Canadian Journal of Microbiology* 35: 169-175.

Fuchs, G., Boll, M. and J. Heider. 2011. Microbial degradation of aromatic compounds - from one strategy to four. *Nature Reviews Microbiology* 9: 803-816.

Garrett, R. M., Pickering, I.J., Haith, C.E. and R. C. Prince. 1998. Photooxidation of crude oils. *Environmental Science & Technology* 32: 3719-3723.

Golyshin, P.N, Chernikova, T.N., Abraham, W.R., Lunsdorf, H., Timmis, K.N. and M.M. Yakimov. 2012. *Oleiphilaceae* fam. nov., to include *Oleiphilus messinensis* gen. nov., sp nov., a novel marine bacterium that obligately utilizes hydrocarbons. *International Journal of Systematic Evolutionary Microbiology* 52: 901-911.

Goodman, R. 2009. Wabamun: A Major Inland Spill. CiteSeerX. http://citeseerx.ist.psu.edu/viewdoc/summary?doi=10.1.1.541.9388. (accessed August 2015).

Government of Canada. 2013. Properties, Composition and Marine Spill Behaviour, Fate and Transport of Two Diluted Bitumen Products from the Canadian Oil Sands. Federal Government Technical Report. Environment Canada.

Greer, C.W., Fortin, N., Sanschagrin, S., Yergeau, E., Whyte, L.G., King, T.L. and K. Lee. 2015. Natural degradation potential for crude oil at sub-zero temperatures in the Canadian Arctic marine environment. Abstract, *38th Arctic and Marine Oilspill Program (AMOP) Technical Seminar.* Environment Canada, Ottawa.

Gros, J., D. Nabi, B. Wurz, L.Y. Wick, C.P.D. Brussaard, J. Huisman, J.R. van der Meer, C.M. Reddy, and J.S. Arey. 2015. First day of an oil spill on the open sea: early mass transfers of hydrocarbons to air and water, Environmental Science and Technology 48 (16): 9400-9411.

Gutierrez, T., Berry, D., Yang, T., Mishamandani, S., Mckay, L. Teske, A. and M.D. Aitken. 2013. Role of bacterial exopolysaccharides (EPS) in the fate of the oil released during the Deepwater Horizon Oil Spill. *PLoS ONE* 8(6): e67717. DOI: 10.1371/journal.pone.0067717.

Haas, C. and S.E.L. Howell. 2015. Ice thickness in the Northwest Passage. *Geophysical Research Letters* 42 (18): 7673-7680.

Harms, H. and T.N.P. Bosma. 1997. Mass transfer limitation of microbial growth and pollutant degradation. *Journal of Industrial Microbiology* 18: 97-105.

Hazen, T.C., Dubinsky, E.A., Desantis, T.Z., Andersen, G.L., Piceno, Y.M., Singh, N., Jansson, J.K., Probst, A., Borglin, S.E., Fortney, J.L., Stringfellow, W.T., Bill, M., Conrad, M.E., Tom, L.M., Chavarria KL, Alusi TR, Lamendella R, Joyner DC, Spier C, Baelum J, Auer M, Zemla ML, Chakraborty, R., Sonnenthal, E.L., D'haeseleer, P., Holman, H.Y., Osman, S., Lu, Z., Van Nostrand, J.D., Deng, Y., Zhou, J. and O.U. Mason. 2010. Deep-sea oil plume enriches indigenous oil- degrading bacteria. *Science* 330: 204-208.

Head, I.M. and R.P.J. Swannell. 1999. Bioremediation of petroleum hydrocarbon contaminants in marine habitats. *Current Opinion in Biotechnology* 10: 234-239.

Head, I.M., D.M. Jones, and W.F.M. Roling. 2006. Marine microorganisms make a meal of oil. *Nature Reviews Microbiology* 4: 173-182.

IPIECA (International Petroleum Industry Environmental Conservation Association). 1995. Biological Impacts of Oil Pollution: Rocky shores. IPIECA Report Series 7:22.

Itoh, S. and T. Suzuki. 1972. Effect of rhamnolipids on growth of a *Pseudomonas aeruginosa* mutant deficient in n-paraffin-utilizing ability. *Agricultural Biology and Chemistry* 36: 2233-2235.

Ivanov, B., Bezgreshnov, A., Kubyshkin, N. and A. Kursheva. 2005. Spreading of oil products in sea ice and their influence on the radiation properties of the snow-ice cover. In: *Proceedings of the 18th International Port and Oceans Engineering Under Arctic Conditions*. pp 853-862.

Johnsen, A. R., Wick, L.Y. and H. Harms. 2005. Principles of microbial PAH-degradation in soil *Environmental Pollution* 133: 71-84.

Jordan, R.E. and J.R. Payne. 1980. Fate and Weathering of Petroleum Spills in the Marine Environment. Ann Arbor Science Publishers, Inc., Ann Arbor, MI.

Jones, T.J., Neustadter, E.L. and K.P. Whittingham. 1978. Water-in-crude oil emulsion stability and emulsion destabilization by chemical demulsifiers. *Journal of Canadian Petroleum Technology* 17: 2-8.

Kanaly, R.A. and S. Harayama. 2010. Advances in the field of high-molecular-weight polycyclic aromatic hydrocarbon biodegradation by bacteria. *Microbial Biotechnology* 3: 136-164.

Karl, T.R., Melillo, J.M.and T.C. Peterson. (eds.). 2009. Global Climate Change Impacts in the United States. A report of the U.S. Global Change Research Program (USGCRP). Cambridge University Press, Cambridge, UK.

Karrick, N.L. 1977. Alteration in petroleum resulting from physical-chemical and microbiological factors. In: Malins, D.C. (Ed) *Effects of Petroleum on Arctic and Subarctic Environments and Organisms Vol. 1. Nature and Fate of Petroleum*. Academic Press, Inc., New York. Pp. 225-299.

Kostka, J.E., Teske, A.P., Joye, S.B. and I.M. Head. 2014. The metabolic pathways and environmental controls of hydrocarbon biodegradation in marine ecosystems. Frontiers in Microbiology: doi: 10.3389/fmicb.2014.00471.

Leahy, J.G. and R.R. Colwell. 1990. Microbial degradation of hydrocarbons in the environment. *Microbiological Reviews* 54: 305-315.

Lea-Smith, D.J., S.J. Biller, M.P. Davey, C.A.R. Cotton, B.M. Perez Sepulveda, A.V. Turchyn, D.J. Scanlan, A.G. Smith, S.M. Chisholm and C.J. Howe. 2015. *Proceedings National Academy of Science:* doi: 10.1073/pnas.1507274112.

ED_001324_00007647-00403

Lee, K., Tremblay, G.H., and Gauthier, J., Cobanli, S.E. and M. Griffin. 1997a Bioaugmentation and biostimulation: a paradox between laboratory and field results. In: *Proceedings of 1997 International Oil Spill Conference*. American Petroleum Institute, Washington DC. Pp. 697-705.

Lee, K., Weise, A.M. and S. St. Pierre. 1997b. Enhanced oil biodegradation with mineral fine interaction. *Spill Science & Technology Bulletin* 3(4): 263-267.

Lee, K., Li, Z., Robinson, B., Kepkay, P., Blouin, M. and B. Doyon. 2011. Field trials of in-situ oil spill countermeasures in ice-infested waters.In:*Proceedings of the 2011 International Oil Spill Conference, Portland, Oregon, USA.May 23 – 26, 2011*.pp. abs160.

Liao, Y., Geng, A. and H. Huang. 2009. The influence of biodegradation on resins and asphaltenes in the Liaohe Basin. *Organic Geochemistry* 40: 312-320.

Margesin, R. and F. Schinner. 2001. Biodegradation and bioremediation of hydrocarbons in extreme environments. *Applied Microbiology Biotechnology* 56(5-6): 650-63.

Margesin R, Moertelmaier, C. and J. Mair. 2013. Low-temperature biodegradation of petroleum hydrocarbons (n-alkanes, phenol, anthracene, pyrene) by four actinobacterial strains. *International Biodeterioration and Biodegradation* 84:185–191.

Martins, L.M. and R.S. Peixoto. 2012. Biodegradation of petroleum hydrocarbons in hypersaline environments. *Brazilian Journal of Microbiology* 43(3): 865-872.

Mbadinga, M., Wang, L-Y., Zhou, L., Liu, J-F., Gu, J-D. and B-Z. Mu. 2011. Microbial communities involved in anaerobic degradation of alkanes. *International Biodeterioration & Biodegradation* 65: 1-13.

McFarlin, K.M., Prince, R.C., Perkins, R. and M.B. Leigh. 2014. Biodegradation of dispersed oil in arctic seawater at -1°C.*PLoS One*9(1): e84297.doi: 10.1371/journal.pone.0084297.

McGenity, T.J., Folwell, B.D., McKew, B.A. and G.O. Sanni. 2012. Marine crude-oil biodegradation: a central role for interspecies interactions. *Aquatic Biosystems* 8: 10.

Minai-Tehrani, D., Minoui, S. and A. Herfatmanesh. 2009. Effect of salinity on biodegradation of polycyclic aromatic hydrocarbons (PAHs) of heavy crude oil in soil. *Bulletin of Environmental Contamination and Toxicology* 82(2): 179-84.

Mitsch, W.J. and J.G. Gosselink. 1993. Wetlands. Van Nostrand Reinhold, New York.

Muskoka Watershed Council. 2007. Muskoka Watersheds Report Card. http://www.muskokawatershed.org/programs/report-card/

Naveenkumar, S., Manoharan, N., Ganesan, S. Manivannan, S.P. and G. Velsamy. 2010. Isolation, screening and in vitro mutational assessment of indigenous soil bacteria for enhanced capability in petroleum degradation. *International Journal of Environmental Science* 1(4): 498.

Noordman, W. H. and D.B. Janssen. 2002. Rhamnolipid stimulates uptake of hydrophobic compounds by *Pseudomonas aeruginosa. Journal of Applied Microbiology* 68: 4502-4508.

NRC (National Research Council). 1985. Oil in the Sea: Inputs, Fates and Effects. National Academies Press, Washington DC.

NRC. 2003. National Research Council: Oil in the Sea III: Inputs, Fates and Effects. National Academies Press, Washington, D.C.

NRC. 2014. Responding to Oil Spills in the U.S. Arctic Marine Environment. National Academies Press, Washington, D.C.

Oberbremer, A. and R. Muller-Hurtig. 1989. Aerobic stepwise hydrocarbon degradation and formation of biosurfactants by an original soil population in a stirred reactor. *Applied Microbiology and Biotechnology* 31: 582-586.

Pérez-Pantoja, D., González, B, and D.H. Pieper. 2010. Aerobic degradation of aromatic hydrocarbons. In: Timmis, K.N., McGenity, T.J., Meer, J.R. and V. Lorenzo (eds.) *Handbook of Hydrocarbon and Lipid Microbiology.* Springer, Berlin Heidelberg. pp. 799-837.

Peters, K.E., C.C. Walters, and J.M. Moldowan. 2005. The biomarker guide. II. Biomarkers and isotopes in petroleum systems and earth history. Cambridge University Press, UK.

ED_001324_00007647-00404

Pineda-Flores, G., Bollarguello, G. and A. Mestahoward. 2004. A microbial mixed culture isolated from a crude oil sample that uses asphaltenes as a carbon and energy source. *Journal of Biodegradation* 15: 145–151.

Polak, J. and B. C-Y. Lu. 1973. Mutual solubility of hydrocarbons and water at 0 and 25°C. *Canadian Journal of Chemistry* 51(12): 4018-4023.

Prince, R. C. 1993. Petroleum spill bioremediation in marine environments. *Critical Reviews in Microbiology* 19: 217-242.

Prince, R.C. 2010. Eukaryotic hydrocarbon degraders. In: Timmis, K.N., McGenity, T.J., van der Meer, J.R. and V. Lorenzo (eds.) *Handbook of Hydrocarbon and Lipid Microbiology*. Springer, Berlin. Pp. 2065-2078.

Prince, R. C. and J. R. Clark. 2004. Bioremediation of marine oil spills. *Studies in Surface Science and Catalysis*151: 495-512.

Pritchard, P.H., Mueller, J.G., Rogers, J.C., Kremer, F.V. and J.A. Glaser. 1992. Oil spill bioremediation: experiences, lessons and results from the *Exxon Valdex* oil spill in Alaska. *Biodegradation* 3: 315-335.

Prosser, J. I., Bohannan, B.J. M., Curtis, T.P., Ellis, R.J., Firestone, M.K. and R.P. Freckleton. 2007. The role of ecological theory in microbial ecology. *Nature Reviews Microbiology* 5: 384–392.

Redfield, A. C. 1934. In: Daniel, R. J. (ed.) *James Johnstone Memorial Volume*. Liverpool University Press, Liverpool. pp. 176–192.

Rojo, F. 2009. Degradation of alkanes by bacteria. *Environmental Microbiology* 11: 2477-2490.

Sabirova, J.S., Ferrer, M., Regenhardt, D., Timmis, K.N. and P.N. Golyshin. 2006. Proteomic insights into metabolic adaptations in *Alcanivorax borkumensis* induced by alkane utilization. *Journal of Bacteriology* 188: 3763-3773.

Sabirova, J.S., Chernikova, T.N., Timmis, K.N. and P.N. Golyshin. 2008. Niche specificity factors of a marine oil-degrading bacterium *Alkanivorax borkumensis* SK2. *FEMS Microbiological Letters* 285: 89-96.

Schneiker, S.V.A., dos Santos, P.M., Bartels, D., Bekel, T., Brecht, M., Buhrmester, J., Chernikova, T.N., Denaro, R., Ferrer, M., Gertler, C., Goesmann, A., Golyshina, O.V., Kaminski, F., Khachane, A.N., Lang, S., Linke, B., McHardy, A.C., Meyer, F., Nechitaylo, T., Pühler, A., Regenhardt, D., Rupp, O., Sabirova, J.S., Selbitschka, W., Yakimov, M.M., Timmis, K.N., Vorhölter, F.J., Weidner, S., Kaiser, O. and P.N. Golyshin. 2006. Genome sequence of the ubiquitous hydrocarbon-degrading marine bacterium *Alcanivorax borkumensis. Nature Biotechnology* 24: 997-1004.

Semanov, G., Volkov, V., Somkin, V. and D. Iljushenko-Krylov. 1997. Working Paper 76, Coastal Pollution Emergency Plan. Part I. International Sea Route Programme (INSROP).

Serova, I. 1992. Behaviour of oil in ice and water at low temperatures. In: *Combatting Marine Oil Spills in Ice and Cold Climates, HELLCOM Seminar, Helsinki, Finland.*

Singsaas, I., Brandvik, P.J., Daling, P.S., Reed, M. and A. Lewis. 1994. Fate and behaviour of oil spilled in the presence of ice - A comparison of the results from recent laboratory, meso-scale flume and field tests. In: *Proceedings of the 17th Arctic and Marine Oilspill Program (AMOP) Technical Seminar. Environment Canada, Ottawa, Ontario*. pp. 355-370.

Short, J.W. and P.M. Harris. 1997. Pristane Monitoring in Mussels and Predators of Juvenile Pink Salmon and Herring, Exxon Valdez Oil Spill Restoration Project Annual Report, http://www.evostc.state.ak.us/Store/AnnualReports/1996-96195-Annual.pdf

Srinivasan, R., Lu, Q., Sorial, G.A. Venosa, A.D. and J. Mullin. 2007. Dispersant effectiveness of heavy fuel oils using the Baffled Flask Test. *Environmental Engineering Science* 24 (9): 1307-1320.

Strassner, J.E. 1968. Effect of pH on Interfacial Films and Stability of Crude Oil-Water Emulsions. *Journal of Petroleum Technology* 20 (3): 303-312.

Teramoto, M., Ohuchi, M., Hatmanti, A., Darmayati, Y., Widyastuti, Y., Harayama, S. and Y. Fukunaga. 2011. *Oleibacter marinus* gen. nov., sp. nov., a bacterium that degrades petroleum aliphatic

ED_001324_00007647-00405

hydrocarbons in a tropical marine environment. *International Journal of Systematic and Evolutionary Microbiology* 61: 375-380.

Tissot, B.P. and D.H. Welte. 1984. Petroleum Formation and Occurrence. SpringerVerlag, Berlin, New York.

Ulrich, A.C., Guigard, S.E., Foght, J.M., Semple, K.M., Pooley K., Armstrong, J.E. and K.W. Biggar. 2009. Effect of salt on aerobic biodegradation of petroleum hydrocarbons in contaminated groundwater. *Biodegradation* 20(1): 27-38.

Uraizee, F.A., Venosa, A.D. and M.T. Suidan. 1998. A model for diffusion controlled bioavailability of crude oil components. *Biodegradation* 8: 287-296.

U.S. EPA. 2013. Climate Impacts on Alaska. http://www.epa.gov/climatechange/impacts-adaptation/alaska.html.

Valentine, D.L., Burch Fisher, G., Bagby, S.C., Nelson, R.K., Reddy, C.M., Sylva, S.P. and M.A. Wood. 2014. Fallout plume of submerged oil from Deepwater Horizon. *Proceedings of the National Academy of Sciences* 111 (145): 15906-15911.

Venkatesh, S., El-Tahan, H., Comfort, G. and R. Abdelnour. 1990. Modeling the behavior of oil spills in Ice-infested waters. *Atmosphere-Ocean* 28: 303-329.

Vinas, M, Grifoli, M., Sabate, J. and A.M. Solanas. 2002. Biodegradation of a crude oil by three microbial consortia of different origins and metabolic capabilities. *Journal of Industrial Microbiology & Biotechnology* 28: 252-260.

Volkering, F., Breure, A. M., Sterkenburg, A and J.G. van Andel. 1992. Microbial degradation of polycyclic aromatic hydrocarbons: E fecf of substrate availability on bacterial growth kinetics. *Applied Microbiology and Biotechnology* 36: 548-522.

Volkering, F., Breure, A.M. and W.H. Rulkens. 1998. Microbiological aspects of surfactant use for biological soil remediation. *Biodegradation* 8: 401-417.

Wang, Z., Fingas, M., Blenkinsopp, S., Sergy, G., Landriault, M., Sigouin, L., Foght, J. and D.W.S. Westlake. 1998. Comparison of oil composition changes due to biodegradation and physical weathering in different oils. *Journal of Chromatography A* 809: 89-107.

Wang, Z., Yang, C., Hollebone, B., Brown, C.E., Landriault, M., Yang, Z. and J. Sun. 2011. Characterization of biodegraded Wabamun spill oil samples incubated with lake fresh sediment or enrichment culture. In: *Proceedings of the 2011 International Oil Spill Conference*. American Petroleum Institute, Washington DC. pp. abs66.http://ioscproceedings.org/doi/pdf/10.7901/2169-3358-2011-1-66

Ward, D.M. and T.D. Brock. 1978. Hydrocarbon biodegradation in hypersaline environments. *Applied Environmental Microbiology* 35: 353-359.

Wenger, L.M., C.L. Davis, and G.H. Isaksen. 2002. Multiple controls on petroleum biodegradation and impact on oil quality. SPE Reservoir Evaluation and Engineering 5: 375-383.

Westlake, D.W.S., Jobson, A., Phillipee, R., and F.D. Cook. 1974. Biodegradability and crude oil composition. *Canadian Journal of Microbiology* 20: 915-928.

Whitehouse, B.G. 1984. The effects of temperature and salinity on the aqueous solubility of polynuclear aromatic hydrocarbons. *Marine Chemistry* 14: 319–332.

Wrenn, B.A., Boufadel, M.C., Suidan, M.T. and A.D. Venosa. 1997a. Nutrient transport during bioremediation of crude oil contaminated beaches.In: *In-Situ and On-Site Bioremediation: Volume 4.* Battelle Press, Columbus, OH. pp. 267-272.

Wrenn, B.A., Suidan, M.T., Strohmeier, K.L., Eberhart, B.L., Wilson, G.J. and A.D. Venosa.1997b. Nutrient transport during bioremediation of contaminated beaches: Evaluation with lithium as a conservative tracer.*Water Research* 31: 515-524.

WSP Canada, Inc. 2014. Risk assessment for marine spills in Canadian waters. Phase 1. Oil spills south of the 60th parallel. Report to Transport Canada. 172 pp. and appendices. http://wcel.org/sites/default/files/file-downloads/131-17593-00_ERA_Oil-Spill-South_150116_pp1-124.pdf

ED_001324_00007647-00406

Yakimov, M.M., Timmis, K.N. and P.N. Golyshin. 2007. Obligate oil-degrading marine bacteria. *Current Opinion in Biotechnology* 18 (3): 257-266.

Yang, S., Wen, X., Zhao, L., Shi, Y, and H. Jin. 2014. Crude oil treatement leads to shift of bacterial communities in soils from the deep active layer and upper permafrost along the China-Russion crude oil pipeline route. *PLoS One* 9(5) e96552.

Zhu, X., Venosa, A.D., Suidan M.T. and K. Lee. 2001. Guidelines for the Bioremediation of Marine Shorelines and Freshwater Wetlands. U.S. Environmental Protection Agency.

Zhu, X., Venosa, A.D., Suidan, M.T. and K. Lee. 2004. Guidelines for the Bioremediation of Oil-Contaminated Salt Marshes. EPA/600/R-04-074. U.S. Environmental Protection Agency.

Zobell, C. E. 1946. Action of microorganisms on hydrocarbons. *Bacteriological Reviews* 10:1–49.

Zobell, C.E. 1973. Microbial degradation of oil: Present state, problems, and perspectives. In: Ahearn, D.G. and S.P. Meyers (eds.) *The Microbial Degradation of Oil Pollutants.* Publication No. LSU-SG-73-01. Louisiana State University, Baton Rouge, LA. Pp. 3-16.

Chapter 4

Ackleh, A.S., Ioup, G.E., Ioup, J.W., Ma, B.L., Newcomb, J.J., Pal, N., Sidorovskaia, N.A. and C. Tiemann. 2012. Assessing the Deepwater Horizon oil spill impact on marine mammal population through acoustics: Endangered sperm whales. *Journal of the Acousti cal Society of America* 131: 2306-2314.

Adams, G.G., Klerks, P.L., Belanger, S.E. and D. Dantin. 1999. The effect of the oil dispersant Omni - Clean® on the toxicity of fuel oil no. 2 in two bioassays with the sheepshead minnow *Cyprinodon variegatus*. *Chemosphere* 39: 2141-2157.

Adams, J., Sweezey, M. and P.V. Hodson. 2014a. Oil and oil dispersant do not cause synergistic toxicity to fish embryos. *Environmental Toxicology and Chemistry* 33: 825-835.

Adams, J.E., Munno, K., Bornstein, J., King, T., Brown, R.S., Ho llebone, B.P. and P.V. Hodson. 2014b. Identification of compounds in heavy fuel oil that are chronically toxic to rainbow trout embryos through effects-driven chemical fractionation. *Environmental Toxicology and Chemistry* 33: 825-835.

ADNR. 2014. Chapter F our: Habitat, Fish, and Wildlife. Alaska Peninsula Areawide Oil and Gas Lease Sales (corrected 12/01/14). Written finding of the director. November 26, 2014. Alaska Department of Natural Resources, Anchorage, AK.

Aeppli, C., Carmichael, C.A., Nelson, R.K., Lemkau, K.L., Graham, W.M., Redmond, M.C., Valentine, D.L. and C.M. Reddy. 2012. Oil weathering after the Deepwater Horizon disaster led to the formation of oxygenated residues. *Environmental Science & Technology* 46(16): 8799-8807.

Alloy, M.M., Boube, I., Griffitt, R.J., Oris, J.T. and A.P. Roberts. 2015. Photo -induced toxicity of Deepwater Horizon slick oil to blue crab (*Callinectes sapidus*) larvae. *Environmental Toxicology and Chemistry* 34: 2061–2066.

Anderson, J.A., Kuhl, A.J. and A.N. Anderson. 2014. Toxicity of oil and dispersed oil on juvenile mud crabs, *Rhithropanopeus harrisii*. *Bulletin of Environmental Contamination and Toxicology* 92: 375-380.

AOSRT (Arctic Oil Spill Response Technology – Joint Industry P rogramme). 2014. Environmental Impacts of Arctic Oil Spills and Arctic Oil Spill Response Technologies. Literature Review and Recommendations. Arctic Oil Spill Response Joint Industry Programme. http://www.arcticresponsetechnology.org/research-projects.

Arias, C.R., Koenders, K. and A.M. Larsen. 2013. Predominant bacteria associated with red snapper from the northern Gulf of Mexico. *Journal of Aquatic Animal Health* 25: 281-289.

Babcock, M. M., Irvine, G. V., Harris, P. M., Cusick, J. A. and S.D. Rice. 1996. Persistence of oiling in mussel beds three and four years after the Exxon Valdez oil spill. In: *Proceedings of the "Exxon Valdez Oil Spill Symposium" held at Anchorage, Alaska, February 2 -5, 1993. American Fisheries Society Symposium. 1996.* American Fisheries Society, Bethesda MD.

Balk, L., Hylland, K., Hansson, T., Berntssen, M.H.G., Beyer, J., Jonsson, G., Melbye, A., Grun g, M., Torstensen, B.E, Borseth, J.F., Skarphedinsdottir, H. and J. Klungsoyr. 2011. Biomarkers in natural fish populations indicate adverse biological effects of offshore oil production. *PLoS ONE* 6: 1-10.

Ballachey, B.E., Bodkin, J.L., Esler, D. and S.D. Rice. 2014. Lessons from the 1989 Exxon Valdez oil spill: A biological perspective. Chap. 9. In: Alford, J.B., Peterson, M.S. and C.C. Green (eds) *Impacts of Oil Spill Disasters on Marine Habitats and Fisheries in North America*.. CRC Press. pp. 181-197.

Barron, M., Carls, M., Short, J. and S. Rice. 2003. Photoenhanced toxicity of aqueous phase and chemically dispersed weathered Alaska North Slope crude oil to Pacific herring eggs and larvae. *Environmental Toxicology and Chemistry* 22: 650-660.

Barron, M.G, and L. Ka'aihue. 2001. Potential for photoenhanced toxicity of spilled oil in Prince William Sound and Gulf of Alaska waters. *Marine Pollution Bulletin* 43: 86-92.

Barron, M.G., Carls, M.G., Heintz, R. and S.D. Rice. 2004. Evaluation of fish early life -stage toxicity models of chronic embryonic exposures to complex polycyclic aromatic hydrocarbon mixtures. *Toxicological Science* 78: 60-67.

Barron, M.G., Carls, M.G., Short, J.W., Rice, S.D., Heintz, R.A., Rau, M. and R.D. Giulio. 2005. Assessment of the phototo xicity of weathered Alaska North Slope crude oil to juvenile pink salmon. *Chemosphere* 60: 105-110.

Barron, M.G., Hemmer, M.J. and C.R. Jackson. 2013. Development of aquatic toxicity benchmarks for oil products using species sensitivity distributions. *Integrated Environmental Assessment and Management* 9: 610-615.

Basu, N., Billiard, S., Fragoso, N., Omoike, A., Tabash, S., Brown, S. and P. Hodson. 2001 Ethoxyresorufin-*o*-de-ethylase induction in trout exposed to mixtures of polycyclic aromatic hydrocarbons. *Environmental Toxicology and Chemistry* 20: 1244-1251.

Bauder, M.B., Palace, V.P. and P.V. Hodson. 2005. Is oxidative stress the mechanism of toxicity in retene - exposed trout larvae? *Environmental Toxicology and Chemistry* 24: 694-702.

Baumann, P.C., Smith, I.R. and C.D. Metcalfe. 1996. Linkages between chemical contaminants and tumors in benthic Great Lakes fish. *Journal of Great Lakes Research* 22: 131-152.

Bell, B., Spotila, J.R. and J. Congdon. 2006. High incidence of deformity in aqu atic turtles in the John Heinz National Wildlife Refuge. *Environmental Pollution* 142: 457-465.

Beyer, J., Trannum, H.C., Bakke, T. and P.V. Hodson. 2015. Environmental effects of the Deepwater Horizon oil spill: a review. *Marine Pollution Bulletin* (submitted).

Billiard, S.M., Querbach, K. and P.V. Hodson. 1999. Toxicity of retene to early life stages of two freshwater fish species. *Environmental Toxicology and Chemistry* 18: 2070-2077.

Billiard, S.M., Hahn, M.E., Franks, D.G., Peterson, R.E., Bols, N.C. and P.V. Hodson. 2002. Binding of polycyclic aromatic hydrocarbons (PAHs) to teleost arylhydrocarbon receptors (AHRs). *Comparative Biochemistry and Physiology B* 133: 55-68.

Billiard, S.M. , Meyer, J.N., Wassenberg, D.M., Hodson, P.V. and R.T. Di Giulio. 2008. Non-additive effects of PAHs on early vertebrate development: mechanisms and implications for risk assessment. *Toxicological Sciences* 105: 5-23.

Birtwell, I.K. 2008. Comments on the ef fects of oil spillage on fish and their habitat - Lake Wabamun, Alberta. Report submitted to the Department of Fisheries and Oceans, Edmonton, AB.

Birtwell, I.K., Fink, R., Brand, D., Alexander, R. and C.D. McAllister. 1999. Survival of pink salmon (*Oncorhynchus gorbuscha*) fry to adulthood following a 10 -day exposure to the aromatic hydrocarbon water -soluble fraction of crude oil and release to the Pacific Ocean. *Canadian Journal of Fisheries and Aquatic Sciences* 56: 2087-2098.

Bolognesi, C., Perrone, E., Roggieri, P. and A. Sciutto. 2006. Bioindicators in monitoring long term genotoxic impact of oil spill: Haven case study. *Marine Environmental Research* 62 (Supplement 1): S287-S291.

Bornstein, J.M., Adams, J.E., Hollebone, B., King, T., Hodson, P.V. and R. S. Brown. 2014. Effects-driven chemical fractionation of heavy fuel oil to isolate compounds toxic to trout embryos. *Environmental Toxicology and Chemistry* 33: 814-824.

ED_001324_00007647-00408

Boudreau, M., Sweezey, M., Lee, K., Hodson, P.V. and S.C. Courtenay. 2009. Toxicity of Orimulsion -400 to early life stages of Atlantic herring (*Clupea harengus*) and mummichog (*Fundulus heteroclitus*). *Environmental Toxicology and Chemistry* 28: 1206-1217.

Brand, D.G., Fink, R., Benge yfield, W., Birtwell, I.K. and C.D. McAllister. 2001. Salt water -acclimated pink salmon fry (*Oncorhynchus gorbusch* a) develop stress -related visceral lesions after 10 -day exposure to sublethal concentrations of the water -soluble fraction of North Slope crud e oil. *Toxicologic Pathology* 29: 574-584.

Brannon, E.L., Quinn, T.P., Whitman, R.P., Nevissi, A.E., Nakatani, R.E. and C.D. McAuliffe. 1986. Homing of adult chinook salmon after brief exposure to whole and dispersed crude oil. *Transactions of the American Fisheries Society* 115: 823-827.

Brannon, E.L., Maki, A.W., Moulton, L.L. and P.R. Parker. 2006. Results from a sixteen year study on the effects of oiling from the Exxon Valdez on adult pink salmon returns. *Marine Pollution Bulletin* 52(8): 892-899.

Brannon, E.L., Collins, K., Cronin, M.A., Moulton, L.L., Maki, A.L. and K.R. Parker. 2012. Review of the Exxon Valdez Oil spill effects on pink salmon in Prince William Sound, Alaska. *Reviews in Fisheries Science* 20(1): 20-60.

Brette, F., Machado, B., Cros, C., I ncardona, J.P., Scholz, N.L. and B.A. Block. 2014. Crude oil impairs cardiac excitation-contraction coupling in fish. *Science* 343: 772-776.

Brewton, R.A., Fulford, R. and R.J. Griffitt. 2013. Gene expression and growth as indicators of effects of the BP De epwater Horizon oil spill on spotted seatrout (*Cynoscion nebulosus*). *Journal of Toxicology and Environmental Health, Part A* 76: 1198-1209.

Brinkworth, L.C., Hodson, P.V., Tabash, S. and P. Lee. 2003. CYP1A induction and blue sac disease in early developmental stages of rainbow trout (*Oncorhynchus mykiss*) exposed to retene. *Journal of Toxicology and Environmental Health - Part A* 66: 47-66.

Brown-Peterson, N.J., Brewton, R.A., Griffitt, R.J. and R.S. Fulford. 2014. Impacts of the Deepwater Horizon oil spill o n the reproductive biology of spotted seatrout (*Cynoscion nebulosus*). Chap. 5. In: Alford, J.B., Peterson, M.S. and C.C. Green (eds) *Impacts of Oil Spill Disasters on Marine Habitats and Fisheries in North America*. CRC Press. pp. 237-252.

Campagna, J.A., Miller, K.W. and S.A. Forman. 2003. Mechanisms of actions of inhaled anesthetics. *New England Journal of Medicine* 348: 2110-2124.

Camus, L. and G.H. Olsen. 2008. Embryo aberrations in sea ice amphipod *Gammarus wilkitzkii* exposed to water soluble fraction of oil. *Marine Environmental Research* 66: 221-222.

Carls, M.G., Rice, S. and J. Hose. 1999. Sensitivity of fish embryos to weathered crude oil: Part I. low -level exposure during incubation causes malformations, genetic da mage, and mortality in larval Pacific herring (*Clupea pallasi*). *Environmental Toxicology and Chemistry* 18: 481-493.

Carls, M.G., Holland, L., Larsen, M., Collier, T.K., Scholz, N.L. and J.P. Incardona. 2008. Fish embryos are damaged by dissolved PAHs, not oil particles. *Aquatic Toxicology* 88(2): 121-127.

Carmack, E.C. and R.W. Macdonald. 2002. Oceanography of the Canadian Shelf of the Beaufort Sea: A setting for marine life. *Arctic* 55(Supplement 1): 29-45.

Carmichael, R.H., Graham, W.M., Aven, A., Worthy, G . and S. Howden. 2012a. Were multiple stressors a 'perfect storm' for northern Gulf of Mexico bottlenose dolphins (*Tursiops truncatus*) in 2011? *Plos One* 7.

Carmichael, R.H., Jones, A.L., Patterson, H.K., Walton, W.C., Perez -Huerta, A., Overton, E.B., Daile y, M. and K.L. Willett. 2012b. Assimilation of oil -derived elements by oysters due to the Deepwater Horizon Oil spill. *Environmental Science & Technology* 46: 12787-12795.

Chase, D.A., Edwards, D.S., Qin, G., Wages, M.R., Willming, M.M., Anderson, T.A. and J.D. Maul. 2013. Bioaccumulation of petroleum hydrocarbons in fiddler crabs (*Uca minax*) exposed to weathered MC-252 crude oil alone and in mixture with an oil dispersant. *Science of the Total Environment* 444: 121–127.

Cleveland, C.J. 2013. Deepwater Horizon oil spill. In: *The Encyclopedia of Earth* . http://www.eoearth.org/view/article/161185/

Clumpner, C. 2015. Oiled Wildlife Preparedness for the Arctic: One Model from Alaska. International Bird Rescue, Astoria, Oregon, USA. Platform Presentation at the 38 th Arctic and Marine Oil Pollution Technical Seminar, June 2-4, 2015, Vancouver, B.C.

Coelho, G., Clark, J. and D. Aurand. 2013. Toxicity testing of dispersed oil requires adherence to standardized protocols to assess potential real world effects. *Environmental Pollution* 177: 185–188.

Colavecchia, M.V., Backus, S.M., Hodson, P.V. and J.L. Parrott. 2004. Toxicity of oil sands to early life stages of fathead minnows (*Pimephales promelas*). *Environmental Toxicology Chemistry* 23: 1709-1718.

Colavecchia, M.V., Hodson, P.V. and J.L. Parrott. 2006. CYP1A induction and blue sac disease on early life stages of white suckers (*Catostomus commersoni*) exposed to oil sands. *Journal of Toxicology and Environmental Health* Part A 69: 1-28.

Colavecchia, M.V., Hodson, P.V. and J.L. Parrott. 2007. The relationships among CYP1A induction, toxicity and eye pathology in early life stages of fish exposed to oil sands. *Journal of Toxicology and Environmental Health* 70: 1542-1555.

Collier, T.K., Singh, S.V., Awasthi, Y.C. and U. Varanasi. 1992. Hepatic xenobiotic metabolizing enzymes in two species of benthic fish showing different prevalences of contaminant - associated liver neoplasms. *Toxicology and Applied Pharmacology* 113: 319-324.

Couillard, C.M., Lee, K., Legare, B. and T.L. King. 2005. Effect of dispersant on the composition of the water-accommodated fraction of crude oil and its toxicity to larval marine fish. *Environmental Toxicology and Chemistry* 24(6): 1496-1504.

Crowe, K.M., Newton, J.C., Kaltenboeck, B. and C. Johnson. 2014. Oxidative stress responses of gulf killifish exposed to hydrocarbons from the Deepwater Horizon oil spill: Potential implications for aquatic food resources. *Environmental Toxicology and Chemistry* 33: 370-374.

de Hoop, L., Schipper, A.M., Leuven, R.S.E.W., Huijbregts, M.A.J., Olsen, G.H., Smit, M.G.D. and A.J. Hendriks. 2011. Sensitivity of polar and temperate marine organisms to oil components. *Environmental Science & Technology* 45(20): 9017-9023.

de Soysa, T.Y., Ulrich, A., Friedrich, T., Pite, D., Compton, S.L., Ok, D., Bernardos, R.L., Downes, G.B., Hsieh, S., Stein, R., Lagdameo, M.C., Halvorsen, K., Kesich, L -R. and M.J.F. Barresi. 2012. Macondo crude oil from the Deepwater Horizon oil spill disrupts specific developmental processes during zebrafish embryogenesis. *BMC Biology* 10: doi:10.1186/1741-7007-10-40.

Debruyn, A.M.H., Wernick, B.G., Stefura, C., McDonald, B.G., Rudolph, B. -L., Patterson, L. and P.M. Chapman. 2007. In situ experimental assessment of lake whitefish development following a freshwater oil spill. *Environmental Science and Technology* 41: 6983 -6989.

de Perre, C., Le Menach, K., Ibalot, F., Parlanti, E. and H. Budzinski. 2014. Development of solid-phase microextraction to study dissolved organic matter-polycyclic aromatic hydrocarbon interactions in aquatic environment. *Analytica Chimica Acta* 807: 51-60.

Dew, W.A., Hontela, A., Rood, S.B. and G.G. Pyle. 2015. Biological effects and toxicity of diluted bitumen and its constituents in freshwater systems. *Journal of Applied Toxicology* 2015: doi: 10.1002/jat.3196.

Di Toro, D.M., McGrath, J.A. and D.J. Hansen. 2000. Technical basis for narcotic chemicals and polycyclic aromatic hydrocarbon criteria. I. Water and tissue. *Environmental Toxicology and Chemistry* 19(8): 1951-1970.

Dubansky, B., Whitehead, A., Miller, J.T., Rice, C.D. and F. Galvez. 2013. Multitissue molecular, genomic, and developmental effects of the Deepwater Horizon Oil Spill on resident Gulf killifish (*Fundulus grandis*). *Environmental Science & Technology* 47: 5074-5082.

Dupuis, A. and F. Ucan-Marin. 2014. A Literature Review on the Aquatic Toxicology of Petroleum Oil: An Overview of Oil Properties and Effects to Aquatic Biota. Fisheries and Oceans Canada, Science Advisory Secretariat, Ottawa, ON.

Dussauze, M., Pichavant -Rafini, K., Le Floch, S., Lemaire, P. and M. Theron. 2015. Acute toxicity of chemically and mechanically dispersed crude oil on juvenile sea bass (*Dicentrarchus labrax*): Absence of synergistic effects between oil and dispersants. *Environmental Toxicology and Chemistry* 34(7): 1543-1551.

Duesterloh, S., Short, J.W. and M.G. Barron. 2002. Photoenhanced toxicity of weathered Alaska North Slope crude oil to the calanoid copepods *Calanus marshallae* and *Metridia okhotensis*. *Environmental Science & Technology* 36: 3953-3959.

Engelhardt, R.F. 1983. Petroleum effects on marine mammals. *Aquatic Toxicology* 4: 199-217.

Fallahtafti, S., Rantanen, T., Brown, R.S., Snieckus, V. and P.V. Hodson. 2012. Toxicity of hydroxylated alkyl-phenanthrenes to the early life stages of Japanese medaka (*Oryzias latipes*). *Aquatic Toxicology* 106-107: 56-64.

ED_001324_00007647-00410

Finch, B.E., Wooten, K.J. and P.N. Smith. 2011. Embryotoxicity of weathered crude oil from the Gulf of Mexico in mallard ducks (*Anas platyrhynchos*). *Environmental Toxicology and Chemist ry* 30: 1885-1891.

Fodrie, F.J. and K.L. Heck, Jr. 2011. Response of coastal fishes to the Gulf of Mexico Oil disaster. *PLoS ONE* 6: e21609.

Fodrie, F.J., Able, K.W., Galvez, F., Heck, K.L., Jensen, O.P., Lopez -Duarte, P.C., Martin, C.W., Turner, R.E. and A. Whitehead. 2014. Integrating organismal and population responses of estuarine fishes in Macondo spill research. *Bioscience* 64: 778-788.

Ford, R.G., Bonnell, M.L., Baroujean, D.H., Page, G.W., Carter, H.W., Sharp, B.E., Heinemann, D. and J.L. Casey. 1996. Total direct mortality of seabirds from the Exxon Valdez oil spill. *American Fisheries Society Symposium* 18: 684-711

Fossi, M.C. and L. Ma rsili. 1997. The use of non -destructive biomarkers in the study of marine mammals. *Biomarkers* 2: 205-216.

Fragoso, N., Hodson, P., Kozin, I., Brown, R. and J. Parrott. 1999. Kinetics of mixed function oxygenase induction and retene excretion in retene -exposed rainbow trout. *Environmental Toxicology and Chemistry* 18: 2268-2274.

Fremling, C.R. 1981. Impacts of a spill of No. 6 fuel oil on Lake Winona. In: *Proceedings of the 1981 Oil Spill Conference*. American Petroleum Institute, Washington, DC. pp. 419-421.

French-McCay, D. P. 2002. Development and application of an oil toxicity and exposure model, OilToxEx. *Environmental Toxicology and Chemistry* 21(10): 2080-2094.

Fry, B. and L.C. Anderson. 2014. Minimal incorporation of Deepwater Horizon oil by estuarine fi lter feeders. *Marine Pollution Bulletin* 80: 282-287.

Garcia, T.I., Shen, Y.J., Crawford, D., Oleksiak, M.F., Whitehead, A.,and R.B. Walter. 2012. RNA -Seq reveals complex genetic response to deepwater horizon oil release in *Fundulus grandis* . *BMC Genomics* 13.

Garcia-Blanco, S., Venosa, A.D., Suidan, M.T., Lee, K., Cobanli, S. and J.R. Haines. 2007. Biostimulation for the treatment of an oil-contaminated coastal salt marsh. *Biodegradation* 18(1): 1-15.

Gardiner, W.W., Word, J.Q., Word, J.D., Perkins, R.A., M cFarlin, K.M., Hester, B.W., Word, L.S.,and C.M. Ray. 2013. The acute toxicity of chemically and physically dispersed crude oil to key arctic species under arctic conditions during the open water season. *Environmental Toxicology and Chemistry* 32: 2284-2300.

Garr, A., Laramore, S.,and W. Krebs. 2014. Toxic effects of oil and dispersant on marine microalgae. *Bulletin of Environmental Contamination and Toxicology* 93: 654-659.

Geiger, H. J., B. G. Bue, B.G., Sharr, S.,Wertheimer, A.C. and T. M. Willette. 1996. A life history approach to estimating damage to Prince William Sound pink salmon caused by the *Exxon Valdez* oil spill. *American Fisheries Society Symposium* 18: 487–498.

Gilfillan, E.S. 1992. Toxic Effects of Oil and Chemically Dispersed Oil on Marine Anima ls and Plants. Prepared for the State of Maine, Department of Environmental Protection.

Gong, Y., Zhao, X., Cai, Z., O'Reilly, S.E., Hao, X. and D. Zhao. 2014. A review of oil, dispersed oil and sediment interactions in the aquatic environment: Influence on the fate, transport and remediation of oil spills. *Marine Pollution Bulletin* 79: 16-33.

Greenberg, B.M., Huang, X.D., Dixon, D.G., Ren, L., McConkey, B.J. and C.L. Duxbury. 1993. Quantitative structure activity relationships for the photo -induced toxicity of polycyclic aromatic hydrocarbons to duckweed - a preliminary model. In: Gorsuch, J.W. , Dwyer, F.J., Ingersoll, C.G. and T.W.L. Point (eds.) *Environmental Toxicology and Risk Assessment* . American Society for Testing and Materials, Philadelphia. pp. 369-378.

Greer, C.D., Hodson, P.V., Li, Z., King, T. and L. Lee. 2012. Toxicity of crude oil chemically dispersed in a wave tank to Atlantic herring (*Clupea harengus*) embryos. *Environmental Toxicology and Chemistry* 31: 1324-1333.

Gulec, I. and D. Holdway. 2000. Toxicity of crude oil and dispersed crude oil to ghost shrimp *Palaemon serenus* and larvae of Australian bass *Macquaria novemaculeata*. *Environmental Toxicology* 15: 91-98.

Häkkinen, J., Vehniäinen, E. and A. Oikari. 2004. High sensitivity of northern pike larvae to UV -B but no UV-photoinduced toxicity of retene. *Aquatic Toxicology* 66: 393-404.

ED_001324_00007647-00411

Haney, J.C., Geiger, H.J. and J.W. Short. 2014a. Bird mortality from the Deepwater Horizon oil spill. I. Exposure probability in the offshore Gulf of Mexico. *Marine Ecology Progress Series* 513: 225-237.

Haney, J.C., Geiger, H.J. and J.W. Short. 2014b. Bird mortality from the Deepwater Horizon oil spill. II. Carcass sampling and exposure probability in the coastal Gulf of Mexico. *Marine Ecology Progress Series* 513: 239-252.

Hatlen, K., Sloan, C.A., Burrows , D.G., Collier, T.K., Scholz, N.L. and J.P. Incardona. 2011. Natural sunlight and residual fuel oils are an acutely lethal combination for fish embryos. *Aquatic Toxicology* 99: 56-64.

Hawkins, S.A., Billiard, S.M., Tabash, S.P., Depew, D.C., Brown, R.S. an d P.V. Hodson. 2002. Altering cytochrome P4501A activity affects polycyclic aromatic hydrocarbon metabolism and toxicity in rainbow trout (*Oncorhynchus mykiss*). *Environmental Toxicology and Chemistry* 21: 1845-1853.

Hedgpeth, B.M. and R.J. Griffitt. 2015. S imultaneous exposure to chronic hypoxia and dissolved PAHs results in reduced egg production and larval survival in the Sheepshead minnow (*Cyprinodon variegatus*). *Environmental Toxicology and Chemistry:* doi: 10.1002/etc.3207.

Heintz, R.A. 2007. Chronic exp osure to polynuclear aromatic hydrocarbons in natal habitats leads to decreased equilibrium size, growth, and stability of pink salmon populations. *Integrated Environmental Assessment and Management* 3: 351-363.

Heintz, R.A., Short, J.W. and S.D. Rice. 1999 . Sensitivity of fish embryos to weathered crude oil: Part II. Increased mortality of pink salmon (*Oncorhynchus gorbuscha*) embryos incubating downstream from weathered Exxon Valdez crude oil. *Environmental Toxicology and Chemistry* 18: 494-503.

Heintz, R.A., Rice, S.D., Wertheimer, A.C., Bradshaw, R.F., Thrower, F.P., Joyce, J.E. and J.W. Short. 2000. Delayed effects on growth and marine survival of pink salmon *Oncorhynchus gorbuscha* after exposure to crude oil during embryonic development. *Marine Ecology Progress Series* 208: 205-216.

Hellebust, A., Hanna, B., Sheath, R.G., Gergis, M. and T.C. Hutchinson. 1975. Experimental crude oil spills on a small subarctic lake in the Mackenzie valley, N.W.T.: Effects on phytoplankton, periphyton, and attache d aquatic vegetation. In: *Proceedings of the 1975 International Oil Spill Conference, March 1975* . American Petroleum Institute, Washington, DC. pp. 509 -515. doi.org/10.7901/2169-3358-1975-1-509

Hemmer, M.J., Barron, M.G. and R.M. Greene. 2011. Comparative toxicity of eight oil dispersants, Louisiana sweet crude oil (LSC), and chemically dispersed LSC to two aquatic test species. *Environmental Toxicology and Chemistry* 30: 2244-2252.

Henkel, J.R., Sigel, B.J. and C.M. Taylor. 2012. Large -scale impacts of the Deepwater Horizon oil spill: Can local disturbance affect distant ecosystems through migratory shorebirds? *Bioscience* 62: 676-685.

Henkel, J.R., Sigel, B.J. and C,M. Taylor. 2014. Oiling rates and condition indices of shorebirds on the northern Gulf of Mexico following the Deepwater Horizon oil spill. *Journal of Field Ornithology* 85: 408-420

Hicken, C.E., Linbo, T.L., Baldwin, D.H., Willis, M.L., Myers, M.S., Holland, L., Larsen, M., Stekoll, M.S., Rice, S.D., Collier, T.K., Scholz, N.L. and J.P. Incardona. 2011. Sublethal exposure to crude oil during embryonic development alters cardiac morphology and reduces aerobic capacity in adult fish. *PNAS* 108(17): 7086-7090.

Highsmith, R. C., Rucker, T. L., Stekoll, M. S., Saupe, S. M., Lindeberg, M. R., Jenne, R. N. and W.P. Erickson. 1996. Impact of the Exxon Valdez oil spill on intertidal biota. In: Proceedings of the "Exxon Valdez Oil Spill Symposium" held at Anchorage, Al aska, February 2 -5, 1993. *American Fisheries Society Symposium. 1996.* American Fisheries Society, Bethesda MD.

Hodson, P.V. and L.L. Williams. 2012. Oil spills from pipelines – why we should be concerned? Platform talk 326 at the 33rd Annual Meeting of the Society of Environmental Toxicology and Chemistry, Long Beach CA, Nov 11-15, 2012.

Hodson, P.V., Ibrahim, I., Zambon, S., Ewert, A. and K. Lee. 2002. Bioavailability to fish of sediment PAH as an indicator of the success of in situ remediation treatments at an experimental oil spill. *Bioremediation Journal* 6: 297-313.

Hodson, P.V., Khan, C.W., Saravanabhavan, G., Clarke, L., Brown, R.S., Hollebone, B., Wang, Z., Short, J.W., Lee, K. and T. King. 2007a. Alkyl PAH in crude oil cause chronic toxicity to early life

ED_001324_00007647-00412

stages of fish. In: *Proceedings of the 30th Arctic and Marine Oilspill Program Technical Seminar*, Edmonton, AB. pp. 291-300.

Hodson, P.V., Qureshi, K., Noble, C.A.J., Akhtar, P. and R.S. Brown. 2007b. Inhibition of CYP1A enzymes by alpha-napthoflavone causes both synergism and antagonism of retene toxicity to rainbow trout (*Oncorhynchus mykiss). Aquatic Toxicology* 81: 275-285.

Hodson, P.V., Collier, T.K. and J.D. Martin. 2011. Technical Data Report: Toxicity to Fish –Potential Effects of an Oil Spill into the Kitimat River from a Northern gateway Pipeline Rupture. Enbridge Northern Gateway Project.

Hollebone, B.P., Fieldhouse, B., Sergey, G., Lamter, P., Wang, Z., Yang, C. and M. Landriaut. 2011. The behaviour of heavy oil in fresh water lakes. In: *Proceedings of the 34th Arctic and Marine Oilspill Program (AMOP) Technical Seminar on Environmental Contamination and Response.* Environment Canada, Ottawa. Pp. 668-709.

Hooten, A. J. and R.C. Highsmith. 1996. Impacts on selected intertidal invertebrates in Herring Bay, Prince William Sound, after the Exxon Valdez oil spill. In: Proceedings of the "Exxon Valdez Oil Spill Symposium" held at Anchorage, Alaska, February 2 -5, 1993. *American Fisheries Society Symposium. 1996.* American Fisheries Society, Bethesda MD.

Incardona, J.P., Collier, T.K. and N.L. Scholz. 2004. Defects in cardiac function precede morphological abnormalities in fish embryos exposed to polycyclic aromatic hydrocarbons. *Toxicology and Applied Pharmacology* 196: 191-205.

Incardona, J.P., Carls, M.G., Teraoka, H., Sloan, C., Collier, T. and N. Scholz. 2005. Aryl hydrocarbon receptor-independent toxicity of weathered crude oil during fish development. *Environmental Health Perspectives* 113: 1755-1762.

Incardona, J.P., Day, H.L., Collier, T.K. and N.L Scholz. 2006. Developmental toxicity of 4-ring polycyclic aromatic hydrocarbons in zebrafish is differentially dependent on AH receptor isoforms and hepatic cytochrome P4501A metabolism. *Toxicology and Applied Pharmacology* 217: 308-321.

Incardona, J.P., Carls, M.G., Day, H.L., Sloan, C.A., Bolton, J.L., Collier, T.K. and N.L. Scholz. 2009. Cardiac arrhythmia is the primary response of embryonic Pacific herring (*Clupea pallasi*) exposed to crude oil during weathering. *Environmental Science &Technology* 43: 201-207.

Incardona, J., Ylitalo, G., Myers, M., Scholz, N., Collier, T., Vines, C., Griffin, F., Smith, E. and G. Cherr. 2011. The 2007 Cosco Busan Oil Spill: Field and Laboratory Assessment of Toxic Injury to Pacific Herring Embryos and Larvae in the San Francisco Estuary. Final Report. NOAA Fisheries, National Oceanic and Atmospheric Administration, Northwest Fisheries Science Center, Environmental Conservation Division, Ecotoxicology and Environmental Assessment Pro grams, Seattle, WA.

Incardona, J.P., Vines, C.A., Anulacion, B.F., Baldwin, D.H., Day, H.L., French, B.L., Labenia, J.S., Linbo, T.L., Myers, M.S., Olson, O.P., Sloan, C.A., Sol, S., Griffin, F.J., Menard, K., Morgan, S.G., West, J.E., Collier, T.K., Ylita lo, G.M., Cherr, G.N. and N.L. Scholz. 2012a. Unexpectedly high mortality in Pacific herring embryos exposed to the 2007 Cosco Busan oil spill in San Francisco Bay. *Proceedings of the National Academy of Sciences* 109(2): E51-E58.

Incardona, J.P., Vines, C. A., Linbo, T.L., Myers, M.S., Sloan, C.A., Anulacion, B.F., Boyd, D., Collier, T.K., Morgan, S., Cherr, G.N. and N.L. Scholz. 2012b. Potent phototoxicity of marine bunker oil to translucent herring embryos after prolonged weathering. *PLoS ONE* 7(2): e30116.

Incardona, J.P., Swarts, T.L., Edmunds, R.C., Linbo, T.L., Aquilina -Beck, A., Sloan, C.A., Gardner, L.D., Block, B.A. and N.L. Scholz. 2013. Exxon Valdez to Deepwater Horizon: Comparable toxicity of both crude oils to fish early life stages. *Aquatic Toxicology* 142-143: 303-316.

Incardona, J.P., Gardner, L.D., Linbo, T.L., Brown, T.L., Esbaugh, A.J., Mager, E.M., Stieglitz, J.D., French, B.L., Labenia, J.S., Laetz, C.A., Tagal, M., Sloan, C.A., Elizur, A., Benetti, D.D., Grosell, M., Block, B.A. and N.L. Scholz. 2014. Deepwater Ho rizon crude oil impacts the developing hearts of large predatory pelagic fish. *Proceedings of the National Academy of Sciences* 111(15): E1510-E1518.

Irons, D. B. 1996. Size and productivity of Black -legged Kittiwake colonies in Prince William Sound before and after the Exxon Valdez oil spill. In: Proceedings of the "Exxon Valdez Oil Spill Symposium" held at Anchorage, Alaska, February 2 -5, 1993. *American Fisheries Society Symposium. 1996.* American Fisheries Society, Bethesda MD.

Judson, R.S., Martin, M.T., Reif, D.M., Houck, K.A., Knudsen, T.B., Rotroff, D.M., Xia, M.H., Sakamuru, S., Huang, R.L., Shinn, P., Austin, C.P., Kavlock, R.J. and D.J. Dix. 2010. Analysis of eight oil spill dispersants using rapid, in vitro tests for endocrine and other biological a ctivity. *Environmental Science & Technology* 44: 5979-5985.

Jung, J., Kim, M., Yim, U.H., Ha, S.Y., An, J.G., Won, J.H., Han, G.M., Kim, N.S., Addison, R.F. and W.J. Shim. 2011. Biomarker responses in pelagic and benthic fish over 1 year following the Hebei Spirit oil spill (Taean, Korea). *Marine Pollution Bulletin* 62: 1859-66.

Jung, J-H., Chae, Y.S., Kim, H.N., Kim, M., Yim, U.H., Ha, S.Y., Han, G.M., An, J.G., Kim, E. and W.J. Shim. 2012. Spatial variability of biochemical responses in resident fish after the M/V Hebei Spirit oil spill (Taean, Korea). *Ocean Science Journal* 47: 209-214.

Jung, J.-H., Hicken, C.E., Boyd, D., Anulacion, B.F., Carls, M.G., Shim, W.J. and J.P. Incardona. 2013. Geologically distinct crude oils cause a common cardiotoxicity syndrom e in developing zebrafish. *Chemosphere* 91: 1146-1155.

Kelly, E.N., Schindler, D.W., Hodson, P.V., Short, J.W., Radmanovich, R. and C.W. Nielsen. 2010. Oil sands development contributes elements toxic at low concentrations in the Athabasca River and its tributaries. *Proceedings of the National Academy of Sciences* 107: 16178-16183.

Kennedy, C.J. and A.P. Farrell. 2008. Immunological alterations in juvenile Pacific herring, *Clupea pallasi*, exposed to aqueous hydrocarbons derived from crude oil. *Environmental Pollution* 153: 638-648.

Khan, R.A. 1991. Influence of concurrent exposure to crude oil and infection with Trypanosoma murmanensis (Protozoa, Mastigophora) on mortality in winter flounder, *Pseudopleuronectes americanus*. *Canadian Journal of Zoology* 69: 876-880.

Kim, M., Yim, U.H., Hong, S.H., Jung, J. -H., Choi, H. -W., An, J., Won, J. and W.J. Shim. 2010. Hebei Spirit oil spill monitored on site by fluorometric detection of residual oil in coastal waters off Taean, Korea. *Marine Pollution Bulletin* 60(3): 383-389.

Kiparissis, Y, Akhtar, P., Hodson, P.V. and R.S. Brown. 2003. Partition Controlled Delivery (PCD) of toxicants: A novel *in vivo* approach for embryotoxicity testing. *Environmental Science & Technology* 37: 2262-2266.

Kocan, R.M., Hose, J.E., Brown, E.D. and T.T. Baker. 1996. Pacific herring (*Clupea pallasi*>) embryo sensitivity to Prudhoe Bay petroleum hydr ocarbons: laboratory evaluation and in situ exposure at oiled and unoiled sites in Prince William Sound. *Canadian Journal of Fisheries and Aquatic Sciences* 53: 2366-2375.

Kuhl, A.J., Nyman, J.A., Kaller, M.D. and C.C. Green. 2013. Dispersant and salinity e ffects on weathering and acute toxicity of South Louisiana crude oil. *Environmental Toxicology and Chemistry* 32: 2611-2620.

Kujawinksi, E.B., Soule, M.C.K., Valentine, D.L., Boysen, A.K., Longnecker, K. and M.C. Redmond. 2011. Fate of dispersants associated with the Deepwater Horizon oil spill. *Environmental Science & Technology* 45: 1298-1306.

Landrum, P.F., Chapman, P.M., Neff , J. and D.S. Page. 2012. Evaluating the aquatic toxicity of complex organic chemical mixtures: Lessons learned from polycyclic aromatic hydrocarbon and petroleum hydrocarbon case studies. *Integrated Environmental Assessment and Management* 8: 217-230.

Laramore, S., Krebs, W. and A. Garr. 2014. Effects of Macondo Canyon 252 oil (naturally and chemically dispersed) on larval *Crassostrea virginica* (Gmelin, 1791). *Journal of Shellfish Research* 33: 709-718.

Lari, E., Abtahi, B., Hashtroudi, M.S., Mohaddes, E. an d K.B. Døving. 2015. The effect of sublethal concentrations of water soluble fraction of crude oil on the chemosensory function of caspian roach, *Rutilus caspicus* (Yakovlev, 1870). *Environmental Toxicology and Chemistry* 34(8):1826-32.

Lavoie, E.J., Tulley -Freiler, L., Bedenko, V. and D. Hoffmann. 1981. Mutagenicity tumor initiating activity, and metabolism of methylphenanthrenes. *Cancer Research* 41: 3441-3447.

Lavoie, E.J., Tulley -Freiler, L., Bedenko, V. and D. Hoffman. 1982. Mutagenic ity of substituted phenanthrenes in *Salmonella typhimurium*. *Mutation Research* 116: 91.

Lee, K. 1983. Vanadium in the aquatic ecosystem. In: Nriagu, J.O. (ed.) *Aquatic Toxicology: Advances in Environmental Science and Technology, Volume 13*. John Wiley & Sons, Inc., N.Y. pp. 155-187.

Lee, K., Wong, C.S., Cretney, W.J., Whitney, F.A., Parsons, T.R., Lalli, C.M. and J. Wu. 1985. Microbial response to crude oil and Corexit 9527: SEAFLUXES enclosure study. *Microbial Ecology* 11: 337-351.

ED_001324_00007647-00414

Lee, L.E.J., Stassen, J. a nd K. Lee. 2000. The mystery snail, *Viviparus georgianus*, as biomonitors of oil -spill and bioremediation strategies in fresh -water habitats. In: *Proceedings of the 27th Annual Aquatic Toxicity Workshop, St. John's, Newfoundland, 1-4 October 2000*. pp. 74.

Lee, K., Venosa, A.D., Suidan, M.T., Garcia -Blanco, S., Greer, C.W., Wohlgeschaffen, G., Cobanli, S.E., Tremblay, G.H. and K.G. Doe. 2003a. Habitat recovery in an oil -contaminated salt marsh following biorestoration treatments. In: *Proceedings of the 2003 International Oil Spill Conference, Vancouver, BC. April 7 -10, 2003* . American Petroleum Institute Publication No. 14730B. pp. 977-982.

Lee, K., R.C. Prince, R.C., Greer, C.W., Doe, K.G., Wilson, J.E.H., Cobanli, S.E., Wohlgeschaffen, G.D., Alroumi, D., King, T. and G.H. Tremblay. 2003b. Composition and toxicity of residual Bunker C fuel oil in intertidal sediments after 30 years. *Spill Science and Technology Bulletin* 8: 187-199.

Lee, K., Boudreau, M., Bugden, J., Burridge, L., Cobanli, S.E., Courtenay, S., Grenon, S., Hollebone, B., Kepkay, P., Li, Z., Lyons, M., Niu, H., King, T.L., MacDonald, S., McIntyre, E.C., Robinson, B., Ryan, S.S. and G. Wohlgeschaffen. 2011. State of Knowledge Review of Fate and Effect of Oil in the Arctic Marine Environment. A repo rt prepared for the National Energy Board of Canada Arctic Roundtable, Fisheries and Oceans Canada. 01/2011; National Energy Board of Canada. https://www.neb-one.gc.ca/ll-eng/livelink.exe/fetch/2000/90463/621169/700096/704342/A2A8R2_-_NEB_State_of_Knowledge_Review_of_Fate_and_Effect_of_Oil_DFO_COOGER.pdf?nodeid=704343&vernum=0.

Lee, K., Budgen, J., Cobanli, S., King, T., McIntyre, C., Robinson, B., Ryan, S. and G. Wohlgeschaffen. 2012. UV -Epifluorescence Microscopy Analysis of Sediments Recovered from the Kalamazoo River. Centre for Offshore Oil, Gas and Energy Research (COOGER), Fisheries and Oceans Canada.

Leighton, F.A. 1993. The toxicity of petroleum oils to birds. *Environmental Reviews* 1: 92-103.

Lewis, M. and R. Pryor. 2013. Toxicities of oils, dispe rsants and dispersed oils to algae and aquatic plants: Review and database value to resource sustainability. *Environmental Pollution* 180: 345-367.

Lin, H., Morandi, G.D., Brown, R.S., Snieckus, V., Rantanen, T., Jørgensen, K.B. and P.V. Hodson. 2015. Quantitative structure -activity relationships for chronic toxicity of alkyl -chrysenes and alkyl -benz[a]anthracenes to Japanese medaka embryos (*Oryzias latipes*). *Aquatic Toxicology* 159: 109-118.

Little, E.E., Hurtubise, R. and L. Cleveland. 1998. Photoenhanced T oxicity of Diluent to the Frog, *Rana sphenocephala.* Columbia Environmental Research Center, Unites States Geological Survey, Columbia, MO.

Little, E., Cleveland, L., Calfee, R. and M. Barron. 2000. Assessment of the photoenhanced toxicity of a weathered oil to the tidewater silverside. *Environmental Toxicology and Chemistry* 19: 926-932.

Litz, J.A., Baran, M.A., Bowen -Stevens, S.R., Carmichael, R.H., Colegrove, K.M., Garrison, L.P., Fire, S.E., Fougeres, E.M., Hardy, R., Holmes, S., Jones , W., Mase -Guthrie, B.E., Odell, D.K., Rosel, P.E., Saliki, J.T., Shannon, D.K., Shippee, S.F., Smith, S.M., Stratton, E.M., Tumlin, M.C., Whitehead, H.R., Worthy, G.A.J. and T.K. Rowles. 2014. Review of historical unusual mortality events (UMEs) in the Gulf of Mexico (1990 -2009): providing context for the multi -year northern Gulf of Mexico cetacean UME declared in 2010. *Diseases of Aquatic Organisms* 112: 161-175.

Logan, K.A., Scott, D., Rosenberger, A. and M. MacDuffee. 2015. Potential Effects on Fraser River Salmon from an Oil Spill by the Trans Mountain Expansion Project. Report p repared for the National Energy Board (NEB) hearings reviewing Kinder Morgan's proposed Trans Mountain Expansion project. Raincoast Conservation Foundation, Sidney, BC. http://www.raincoast.org/wp-content/uploads/2015/06/RCF-Fraser-SS-Salmon-NEB-May-2015.pdf

Longpré, D., Lee, K., Tremblay, G.H. and V. Jarry. 2000. The response of *Scirpus pungens* to crude oil contaminated sediments. In: *Proceedings of the 27th Annual Aquatic Toxicity Workshop, St. John's, Newfoundland, 1-4 October 2000*. pp. 127.

Loughlin, T.R., Ballachey, B.E. and B.A. Wright. 1996. Overview of studies to determine injury caused b y the Exxon Valdez oil spill to marine mammals. In: *Proceedings of the "Exxon Valdez Oil Spill*

Symposium" held at Anchorage, Alaska, February 2 -5, 1993 . American Fisheries Society, Bethesda, MD. pp. 798-808.

Madison, B.N., Hodson, P.V. and V.S. Langlois. 2 015. Diluted bitumen causes deformities and molecular responses indicative of oxidative stress in Japanese medaka embryos *Aquatic Toxicology* 165: 222-230.

Mager, E.M., Esbaugh, A.J., Stieglitz, J.D., Hoenig, R., Bodinier, C., Incardona, J.P., Scholz, N.L., Benetti, D.D. and M. Grosell. 2014. Acute embryonic or juvenile exposure to Deepwater Horizon Crude Oil impairs the swimming performance of mahi -mahi (*Coryphaena hippurus*). *Environmental Science & Technology* 48: 7053-7061.

Martin, J.D., Adams, A., Hollebone, B.P., King, T., Brown, R.S. and P.V. Hodson. 2014. Chronic toxicity of heavy fuel oils to fish embryos using multiple exposure scenarios. *Environmental Toxicology and Chemistry* 33: 677-687.

Marty, G.D., Short, J.W., Damb ach, D.M., Willits, N.H., Heintz, R.A., Rice, S.D., Stegeman, J.J. and D.E. Hinton. 1997. Ascites, premature emergence, increased gonadal cell apoptosis, and cytochrome P4501A induction in pink salmon larvae continuously exposed to oil -contaminated gravel during development. *Canadian Journal of Zoology* 75: 989-1007.

Marty, G.D., Quinn, T.J., Carpenter, G., Meyers, T.R. and N.H. Willits. 2003. Role of disease in abundance of a Pacific herring (*Clupea pallasi*) population. *Canadian Journal of Fisheries and Aqu atic Sciences* 60: 1258-1265.

Mathew, R., McGrath, J.A. and D.M. Di Toro. 2008. Modeling polycyclic aromatic hydrocarbon bioaccumulation and metabolism in time -variable early life state exposures. *Environmental Toxicology and Chemistry* 27: 1515-1525

McCarty, L.S. and D. Mackay. 1993. Enhancing ecotoxicological modeling and assessment: Body residues and modes of toxic action. *Environmental Science & Technology* 27: 1719-1728.

McGrath, J.A. and D.M. Di Toro. 2009. Validation of the target lipid model for toxici ty assessment of residual petroleum constituents: monocyclic and polycyclic aromatic hydrocarbons. *Environmental Toxicology and Chemistry* 28: 1130-1148.

McIntosh, S., King, T., Wu, D. and P.V. Hodson. 2010. Toxicity of dispersed crude oil to early life sta ges of Atlantic herring (*Clupea harengus*). *Environmental Toxicology and Chemistry* 29: 1160-1167.

Melbye, A.G., Brakstad, O.G., Hokstad, J.N., Gregersen, I.K., Hansen, B.H., Booth, A.M., Rowland, S.J. and K.E. Tollefsen. 2009. Chemical and toxicological ch aracterization of an unresolved complex mixture-rich biodegraded oil. *Environmental Toxicology and Chemistry* 28(9): 1815-1824.

Middaugh, D.P., Chapman, P.J., Shelton, M.E., McKenney, C.L. and L.A. Courtney. 2002. Effects of fractions from biodegraded Alask a North Slope crude oil on embryonic inland silversides, *Menidia beryllina*. *Archives of Environmental Contamination and Toxicology* 42: 236-243.

Misaki, K., Kawami, H., Tanaka, T., Handa, Y., Nakamura, M., Matsui, S., and T. Matsuda. 2007. Aryl hydrocarbon receptor ligand activity of polycyclic aromatic ketones and polycyclic aromatic quinones. *Environmental Toxicology and Chemistry* 26: 1370-1379.

Mitchelmore, C.L. and J.E. Baker. 2010. Acute and chronic effects of oil dipsersant and dispersed oil to sensitive symbiotic cnidarian species, including corals. Final Project Report to the Coastal Response Research Center. University of Maryland, Center for Environmental Science, Solomons, MD.

Moody, R.M., Cebrian, J. and K.L. Heck. 2013. Interannual recruitment dynamics for resident and transient marsh species: Evidence for a lack of impact by the Macondo oil spill. *Plos One* 8.

Muhling, B.A., Roffer, M.A., Lamkin, J.T., Ingram, G.W., Upton, M. A., Gawlikowski, G., Muller-Karger, F., Habtes, S. and W.J. Richards. 2012. Overlap between Atlantic bluefin tuna spawning grounds and observed Deepwater Horizon surface oil in the northern Gulf of Mexico. *Marine Pollution Bulletin* 64: 679-687.

Myers, M.S., Johnson, L.L. and T.K. Collier. 2003. Establishing the causal relationship between polycyclic aromatic hydrocarbon (PAH) exposure and hepatic neoplasms and neoplasia -related liver lesions in English sole (*Pleuronectes vetulus*). *Human and Ecological Risk Assessment* 9: 67-94.

Nahrgang, J., Camus, L., Carls, M.G., Gonzalez, P., Jonsson, M., Taban, I.C., Bechmann, R.K., Christiansen, J.S. and H. Hop. 2010. Biomarker responses in polar cod (*Boreogadus saida*) exposed to the water soluble fraction of crude oil. *Aquatic Toxicology* 97(3): 234-242.

ED_001324_00007647-00416

Neff, J., Ostazeski, S., Gardiner, W. and I. Stejskal. 2000. Effects of weathering on the toxicity of three offshore Australian crude oils and a diesel fuel to marine animals. *Environmental Toxicology and Chemistry* 19: 1809-1821.

Neff, J.M., Page, D.S., Landrum, P.F. and P.M. Chapman. 2013. The importance of both potency and mechanism in dose -response analysis: An example from exposure of Pacific herring (Clupea pallasi) embryos to low concentrations of weathered crude oil. *Marine Pollution Bulletin* 67: 7-15.

Nordtug, T., Olsen, A.J., Altin, D., Meier, S., Overrein, I., Hansen, B.H., and A. Johansen. 2011a. Method for generating parameterized ecotoxicity data of dispersed oil for use in environmental modelling. *Marine Pollution Bulletin* 62: 2106-2113.

Nordtug, T., Olsen, A.J., Altin, D., Overrein, I., Storoy, W., Hansen, B.H., and F. De Laender. 2011b. Oil droplets do not affect assimilation and survival probability of first feeding larvae of North -East Arctic cod. *Science of the Total Environment* 412-413: 148-153.

Nordtug, T., Olsen, A.J., Salaberria, I., Øverjordet, I.B., Altin, D., Størdal, I.F. and B.H. Hansen. 2015. Oil droplet ingestion and oil fouling in the copepod *Calanus finmarchicus* exposed to mechanically and chemically dispersed crude oil. *Environmental Toxicology and Chemistry* 34(8): 1899-1906.

NRC (National Research Council). 2005. Toxicological effects of dispersants and dispersed oil. In: *Oil Spill Dispersants. Efficacy and Effects*. National Academies Press, Washington, DC. pp. 193-275.

NRC. 2013. An Ecosystem Services Approach to Assessing the Impacts of the Deepwater Horizon Oil Spill in the Gulf of Mexico. National Academies Press Washington, DC.

Oakley, K. L. and K.J. Kuletz. 1996. Population, reproduction, and fora ging of pigeon guillemots at Naked Island, Alaska, before and after the Exxon Valdez oil spill. In: *Proceedings of the "Exxon Valdez Oil Spill Symposium" held at Anchorage, Alaska, February 2-5, 1993. American Fisheries Society Symposium. 1996.* American Fisheries Society, Bethesda MD.

Olsen, G.H., Klok, C., Hendriks, A.J., Geraudie, P., De Hoop, L., De Laender, F., Farmen, E., Grøsvik, B.E., Hansen, B.H., Hjorth, M., Jansen, C.R., Nordtug, T., Ravagnan, E., Viaene, K. and J. Carroll. 2013. Toxic ity data for modeling impacts of oil components in an Arctic ecosystem. *Marine Environmental Research* 90: 9-17.

Olsvik, P.l.A., Hansen, B.r.H., Nordtug, T., Moren, M., Holen, E., and K.K. Lie. 2011. Transcriptional evidence for low contribution of oil drop lets to acute toxicity from dispersed oil in first feeding Atlantic cod (Gadus morhua) larvae. *Comparative Biochemistry and Physiology Part C: Toxicology & Pharmacology* 154(4): 333-345.

Olsvik, P., Lie, K., Nordtug, T., and B.H. Hansen. 2012. Is chemically dispersed oil more toxic to Atlantic cod (Gadus morhua) larvae than mechanically dispersed oil? A transcriptional evaluation. *BMC Genomics* 13(1): 702.

Owens, E.H., Sienkiewicz, A.M. and G.A. Sergy. 1999, Evaluation of Shoreline Cleaning Versus Natural Recovery: The Metula Spill and the Komi Operations. In: *Proceedings of the 1999 International Oil Spill Conference*. American Petroleum Institute, Washington, DC. Pp. 503-509.

Parkerton, T.F., Stone, M.A. and D.J. Letinski. 2000. Assessing the aquatic toxicity of complex hydrocarbon mixtures using solid phase microextraction. *Toxicology Letters* 112–113: 273–282.

Pearson, W.H., Elston, R.A., Bienert, R.W., Drum, A.S. and L.D. Antrim. 1999. Why did the Prince William Sound, Alaska, Pacific herring (*Clupea pallasi*) fisheries collapse in 1993 and 1994? Review of hypotheses. *Canadian Journal of Fisheries and Aquatic Sciences* 56: 711-737.

Perkins, R.A., Rhoton, S., and C. Behr -Andres. 2005. Comparative marine toxicity testing: A cold -water species and standard warm -water test species exposed to crude oil and dispersant. *Cold Regions Science and Technology* 42(3): 226-236.

Place, B., Anderson, B., Mekebri, A., Furlong, E.T., Gray, J.L., Tjeerdema, R., and J. Field. 2010. A role for analytical chemistry in advancing our u nderstanding of the occurrence, fate, and effects of Corexit oil dispersants. *Environmental Science & Technology* 44: 6016-6018.

Piatt, J. F. and P. Anderson. 1996. Response of common murres to the Exxon Valdez oil spill and long -term changes in the Gulf of Alaska marine ecosystem. In: *Proceedings of the "Exxon Valdez Oil Spill Symposium" held at Anchorage, Alaska, February 2 -5, 1993*. American Fisheries Society, Bethesda, MD. pp. 720-737

Pilcher, W., Miles, S., Tang, S., Mayer, G., and A. Whitehead. 2014. Ge nomic and genotoxic responses to controlled weathered-oil exposures confirm and extend field studies on impacts of the Deepwater Horizon oil spill on native killifish. *PLoS ONE* 9: e106351.

ED_001324_00007647-00417

Pollino, C.A. and D.A. Holdway. 2002. Toxicity testing of crude oil and related compounds using early life stages of the crimson -spotted rainbowfish (*Melanotaenia fluviatilis*). *Ecotoxicology and Environmental Safety* 52: 180-189.

Raimondo, S., Jackson, C.R., Krzykwa, J., Hemmer, B.L., Awkerman, J.A., and M.G. Barron. 2014. Developmental toxicity of Louisiana crude oil -spiked sediment to zebrafish. *Ecotoxicology and Environmental Safety* 108: 265-272.

Ramachandran, S.D., Hodson, P.V., Khan, C.W. and K. Lee. 2004a. Oil dispersant increases PAH uptake by fish exposed to crude oil. *Ecotoxicology and Environmental Safety* 59: 300-308.

Ramachandran, S.D., Khan, C.W. and P.V. Hodson. 2004b. Role of droplets in promoting up take of PAHs by fish exposed to chemically dispersed crude oil. In: *Proceedings of the Twenty -Seventh Arctic and Marine Oilspill Program (AMOP) Technical Seminar Edmonton, AB* . Environment Canada, Ottawa, ON. pp. 765–772.

Ramachandran, S.D., Sweezey, M.J., Hodson, P.V., Boudreau, M., Courtenay, S., King, T., Dixon, J.A. and K. Lee. 2006. Influence of salinity and fish species on PAH uptake from dispersed MESA crude oil. *Marine Pollution Bulletin* 52: 1182-1189.

Räsänen, K.M., Vehniäinen, E. -R., and A.O.J. Oik ari. 2011. Different sensitivities of whitefish (*Coregonius lavaretus*) and northern pike (*Esox lucius*) eleutheroembryos to photoinduced toxicity of polycyclic aromatic hydrocarbons. *Polycyclic Aromatic Hydrocarbons* 31: 65-83.

Redman, A.D. 2015. Role of entrained droplet oil on the bioavailability of petroleum substances in aqueous exposures. *Marine Pollution Bulletin* 97: 342-348.

Redman, A.D., McGrath, J.A., Stubblefield, W.A., Maki, A.W. and D.M. Di Toro. 2012. Quantifying the concentration of crude oil mi crodroplets in oil–water preparations. *Environmental Toxicology and Chemistry* 31(8): 1814-1822.

Redman, A.D., Parkerton, T.F., Letinski, D.J., Manning, R.G., Adams, J.E. and P.V. Hodson. 2014. Evaluating toxicity of heavy fuel oil fractions using complimen tary modeling and biomimetic extraction methods. *Environmental Toxicology and Chemistry* 33: 2094-2104.

Redman, A.D., and T.F. Parkerton. 2015. Guidance for improving comparability and relevance of oil toxicity tests. *Marine Pollution Bulletin* 98 (1-2): 156-170.

Ren, L., Huang, X.D., MacConkey, B.J., Dixon, D.G., and B.M. Greenberg. 1994. Photoinduced toxicity of three polycyclic aromatic hydrocarbons (Fluoranthene, pyrene, and naphthalene) to the duckweed *Lemna gibba* L G-3. *Ecotoxicology and Environmental Safety* 28: 160-171.

Reynaud, S. and P. Deschaux. 2006. The effects of polycyclic aromatic hydrocarbons on the immune system of fish: A review. *Aquatic Toxicology* 77: 229-238.

Rhodes, S., Farwell, A., Hewitt, L.M., MacKinnon, M., and D.G. Dixon. 2005. The ef fects of dimethylated and alkylated polycyclic aromatic hydrocarbons on the embryonic development of the Japanese medaka. *Ecotoxicology and Environmental Safety* 60: 247-258.

Rico-Martinez, R., Snell, T.W., and T.L. Shearer. 2013. Synergistic toxicity of Ma condo crude oil and dispersant Corexit 9500A® to the *Brachionus plicatilis* species complex (Rotifera). *Environmental Pollution* 173: 5-10.

Rood, S. and E. Hillman. 2013. The 2012 Red Deer River oil spill: Analyzing environmental impacts in the floodplain zone. *Western Canadian Spill Services Ltd Newsletter* 2(8): 4.

Rowe, C.L., Mitchelmore, C.L., and J.E. Baker. 2009. Lack of biological effects of water accommodated fractions of chemically - and physically -dispersed oil on molecular, physiological, and behavioral traits of juvenile snapping turtles following embryonic exposure. *Science of the Total Environment* 407(20): 5344-5355.

Rozas, L., Minello, T. and M.S. Miles. 2014. Effect of Deepwater Horizon Oil on growth rates of juvenile penaeid shrimps. *Estuaries and Coasts* 37: 1403-1414.

RPI (Research Planning, Inc.). 1991 . Sea Turtles and Oil –A Synopsis of the Available Literature. RPI/R/91/10/14-9. Prepared for National Oceanic and Atmospheric Administration, Seattle, WA.

Saba, V.S., and J.R. Spotila. 2003. Survival and behavior of freshwater turtles after rehabilitation from an oil spill. *Environmental Pollution* 126: 213-223.

Schein, A., Scott, J.A., Mos, L. and P.V. Hodson. 2009. Oil dispersion increases the apparent bioavailability and toxicity of diesel to rainbow trout (*Oncorhynchus mykiss*). *Environmental Toxicology and Chemistry* 28: 595-602.

Schwacke, L., Smith, C., Townsend, F., Wells, R., Hart, L., Balme, B., Collier, T., De Guise, S., Fry, M., Guillette Jr., L., Lamb , S., Lane, S., McFee , W., Place , N., Tumlin, M., Ylitalo, G., Zolman, E.,

ED_001324_00007647-00418

and T. Rowles. 2014. Health of common bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana, following the *Deepwater Horizon* Oil Spill. *Environmental Science & Technology* 48: 93-103.

Scott, J.A. and P.V. Hodson. 2008. Evidence for multiple mechanisms of toxicity in larval rainbow trout (*Oncorhynchus mykiss*) co-treated with retene and a -naphthoflavone. *Aquatic Toxicology* 88: 200-206.

Scott, J.A., Ross, M., Lemire, B.C. and P.V. Hodson. 2009. Embryotoxicity of retene in cotreatment with 2-aminoanthracene, a cytochr ome P4501A inhibitor, in rainbow trout (*Oncorhynchus mykiss*). *Environmental Toxicology and Chemistry* 28: 1304-1310.

Scott, J.A., Incardona, J.P., Pelkki, K., Shepardson, S. and P.V. Hodson. 2011. AhR2 -mediated, CYP1A-independent cardiovascular toxicity in zebrafish (*Danio rerio*) embryos exposed to retene. *Aquatic Toxicology* 101: 165–174.

Sellin Jeffries, M.K., Claytor, C., Stubblefield, W., Pearson, W.H. and J.T. Oris. 2013. Quantitative risk model for polycyclic aromatic hydrocarbon photoinduced toxicity in Pacific herring following the Exxon Valdez oil spill. *Environmental Science & Technology* 47: 5450-5458.

Sergy, G.A. and P.J. Blackhall. 1987. Conclusions of the Baffin Island oil spill project. *Arctic* 40: 1-Q.

Sergy, G.A. and E.H. Owens. 2011. Differences and similarities in freshwater and marine shoreline oil spill response. In: *Proceedings of the 2011 International Oil Spill Conference.* p. abs62.

Shafir, S., VanRijn, J. and B. Rinkevich. 2007. Short and long term toxicity of crude oil and oil dispersants to two representative coral species. *Environmental Science & Technology* 41: 5571-5574.

Shen, A., Tang, F. and X. Shen. 2010. Toxicity of crude oil and fuel oil on the embryonic development of the large yellow croaker (*Larmichthys crocea*). In: *Proceedings of 3rd International Conference on Biomedical Engineering and informatics, Oct 16-18, 2010. Yantai, China.*

Sharp, B. E., Cody, M. and R. Turner. 19 96. Effects of the Exxon Valdez oil spill on the black oystercatcher. In: *Proceedings of the "Exxon Valdez Oil Spill Symposium" held at Anchorage, Alaska, February 2-5, 1993*. American Fisheries Society, Bethesda MD.

Short, J.W. 2008. An Evaluation of Oil F ate, Persistence and Toxic Potential to Fish Following the August 2005 Canadian National Derailment Spill into Lake Wabamun, Alberta. August 27, 2008. https://docs.neb-one.gc.ca/ll-eng/llisapi.dll/Open/2586376.

Short, J.W. 2015. The behaviour and environmental impacts of crude oil released into aqueous environments. Presentation to the Royal Society of Canada Expert Panel on Oil in Water, Halifax, NS, April 10, 2015.

Shugart, L.R. 1 995. Chap 13. Environmental genotoxicology. In: Rand, G.M. (Ed) *Fundamentals of Aquatic Toxicology. Effects, Environmental Fate, and Risk Assessment.* 2nd ed.. Taylor and Francis, Washington, DC. pp 405-419.

Sinclair, J. 2012. Appendix 2. Critique of the exposure and hazard assessment for river otters (*Lutra lutra*). In: Hodson, P.V. and J. Martin (eds.) *Review for the Haisla Nation Council, Kitimaat Village, BC, of the Technical Data Report "Ecological and Human Health Risk Assessment for Pipeline Spills" by Enbridge Northern Gateway Project*". pp 46-52.

Singer, M.M., Smalheer, D.L., Tjeerdema, R.S., and M. Martin. 1991. Effects of spiked exposure to an oil dispersant on the early life stages of four marine species. *Environmental Toxicology and Chemistry* 10(10): 1367-1374.

Singer, M.M., Aurand, D., Bragin, G.E., Clark, J.R., Coelho, G.M., Sowby, M.L. and R.S. Tjeerdema. 2000. Standardization of the preparation and quantitation of water -accommodated fractions of petroleum for toxicity testing. *Marine Pollution Bulletin* 40(11): 1007-1016.

Sørhus, E., Edvardsen, R.B., Karlsen, Ø., Nordtug, T., van der Meeren, T., Thorsen, A., Harman, C., Jentoft, S. and S. Meier. 2015. Unexpected interaction with dispersed crude oil droplets drives severe toxicity in Atlantic haddock embryos. *PLoS ONE* 10(4): e0124376.

Stagg, R.M., Rusin, J., McPhail, M.E., McIntosh, A.D., Moffat, C.F. and J.A. Craft. 2000. Effects of polycyclic aromatic hydrocarbons on expression of cyp1a in salmon (*Salmo salar)* following experimental exposure and after the Braer oil spill. *Environmental Toxicology and Chemistry* 19: 2797-2805.

Stout, S.A. 2015. Chemical Fingerprinting Assessment of Exposure of Dolphins to Macondo Oil During and After the Deepwater Horizon Oil Spill. DWH NRDA Chemistry Technical Wor king Group Report (September 1, 2015)

Szedlmayer, S.T. and P.A. Mudrak. 2014. Influence of age -1 conspecifics, sediment type, dissolved oxygen, and the Deepwater Horizon oil spill on recruitment of age -0 red snapper in the northeast Gulf of Mexico during 2 010 and 2011. *North American Journal of Fisheries Management* 34: 443-452.

Tamis, J.E., Jongbloed, R.H., Karman, C.C., Koops, W. and A.J. Murk. 2012. Rational application of chemicals in response to oil spills may reduce environmental damage. *Integrated Environmental Assessment and Management* 8: 231-241.

Teal, J.M. and R.W. Howarth. 1984. Oil spill studies: A review of ecological effects. *Environmental Management* 8: 27-44.

Thormann, M.N. and S.E. Bayley. 2008. Impacts of the CN Rail Oil Spill on S oftstem Bulrush-Dominated Lacustrine Marshes in Wabamun Lake. Report submitted to Alberta Environment, Edmonton, AB.

Thorne, R.E. and G.L. Thomas. 2008. Herring and the "Exxon Valdez" oil spill: An investigation into historical data conflicts. *ICES Journal of Marine Sciences* 65: 44-50.

Tollefsen, K.E., Petersen, K. and S.J. Rowland. 2012. Toxicity of synthetic naphthenic acids and mixtures of these to fish liver cells. *Environmental Science & Technology* 46: 5143-4150.

Toor, N.S., E.D. Franza,, P.M. Fedorak, M.D. M acKinnon, and L. Karsten. 2013. Degradation and aquatic toxicity of naphthenic acids in oil sands process -affected waters using simulated wetlands. *Chemosphere* 90: 449-458.

Tran, T., Yazdanparast, A. and E.A. Suess. 2014. Effect of oil spill on birds: a gr aphical assay of the Deepwater Horizon oil spill's impact on birds. *Computational Statistics* 29: 133-140.

Truscott, B., Walsh, J.M., Burton, M.P., Payne, J.F. and D.R. Idler. 1983. Effect of acute exposure to crude petroleum on some reproductive hormones i n salmon and flounder. *Comparative Biochemistry and Physiology Part C: Toxicology* 75: 121-130.

Turcotte, D., Akhtar, P., Bowerman, M., Kiparissis, Y., Brown, R.S. and P.V. Hodson. 2011. Measuring the toxicity of alkyl-phenanthrenes to early life stages of medaka (*Oryzias latipes)* using partition controlled delivery. *Environmental Toxicology and Chemistry* 30: 487–495.

USFWS. 2015. Draft Damage Assessment and Restoration Plan/Environmental Assessment for the July 25-26, 2010 Enbridge Line 6B Oil Discharges near Marshall, MI. U.S. Fish and Wildlife Service, Nottawaseppi Huron Band of the Potawatomi Tribe, Match -E-Be-Nash-She-Wish Band of the Pottawatomi Indians, National Oceanic and Atmospheric Administration, Michigan Department of Environmental Quality, and Michigan Department of Natural Resources.

Valentine, D.L., Burch Fisher, G., Bagby, S.C., Nelson, R.K., Reddy, C.M., Sylva, S.P. and M.A. Woo. 2014. Fallout plume of submerged oil from Deepwater Horizon. *Proceedings of the National Academy of Sciences* 111(45): 15906-15911

van der Ham, J.L. and K. de Mutsert. 2014. Abundance and size of Gulf shrimp in Louisiana's coastal estuaries following the Deepwater Horizon Oil spill. *PLoS ONE* 9: e108884.

Van Tamelen, P. G. and M.S. Stekoll. 1996. Population response o f the brown alga *Fucus gardneri* and other algae in Herring Bay, Prince William Sound, to the *Exxon Valdez* oil spill. In: *Proceedings of the "Exxon Valdez Oil Spill Symposium" held at Anchorage, Alaska, February 2 -5, 1993*. American Fisheries Society, Bethesda MD.

Vehniainen, E.R., Hakkinen, J. and A. Oikari. 2003. Photoinduced lethal and sublethal toxicityof retene, a polycyclic aromatic hydrocarbon derived from resin acid, to coregonid larvae. *Environmental Toxicology and Chemistry* 22: 2995-3000.

Venn-Watson, S., Colegrove, K.M., Litz, J., Kinsel, M., Terio, K., Saliki, J., Fire, S., Carmichael, R., Chevis, C., Hatchett, W., Pitchford, J., Tumlin, M., Field, C., Smith, S., Ewing, R., Fauquier, D., Lovewell, G., Whitehead, H., Rotstein, D., McFee, W., F ougeres, E. and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico Common Bottlenose Dolphins (*Tursiops truncatus*) found dead following the Deepwater Horizon oil spill. *PLoS ONE* 10(5): e0126538.

Venosa, A.D., Lee, K., Suidan, M.T., Garcia -Blanco, S. Cobanli, S., Moteleb, M., Haines, J.R., Tremblay, G. and M. Hazelwood. 2002. Bioremediation and biorestoration of crude oil -contaminated freshwater wetland on the St. Lawrence River. *Bioremediation Journal* 6: 261-281.

Vikebø, F.B., Rønningen, P., Meier, S., Grøsvik, B.E. and V.S. Lien. 2015. Dispersants have limited effects on exposure rates of oil spills on fish eggs and larvae in Shelf seas. *Environmental Science & Technology* 49: 6061–6069

ED_001324_00007647-00420

Vrabie, C.M., Candido, A., van Duursen, M.B.M. and M.T.O. Jonker. 2010. Specific *in vitro* toxicity of crude and refined petroleum products: II. Estrogen (α and β) and androgen receptor -mediated responses in yeast assays. *Environmental Toxicology and Chemistry* 29(7): 1529-1536.

Vrabie, C.M., Candido, A., van den Berg, H., Murk, A.J., van Duursen, M.B.M. and M.T.O. Jonker. 2011. Specific in vitro toxicity of crude and refined petroleum products: 3. Estrogenic responses in mammalian assays. *Environmental Toxicology and Chemistry* 30(4): 973-980.

Vrabie, C.M., Sinnige, T.L., Murk, A.J. and M.T.O. Jonker. 2012. Effect -directed assessment of the bioaccumulation potential and chemical nature of ah receptor agonists in crude and refined oils. *Environmental Science & Technology* 46(3): 1572-1580.

Wang, Y., Zhou, Q., Peng, S., Ma, L. and X. Niu. 2009. Toxic effects of crude -oil-contaminated soil in aquatic environment on *Carassius auratus* and their hepatic antioxidant defense system. *Journal of Environmental Sciences* 21: 612-617.

Wang, Z., Hollebone, B. P., Fingas, M., Fieldhouse, B., Sigouin, L., Landriault, M., Smith, P., Noonan, J., Thouin, G. and J.W. Weaver. 2003. Characteristics of Spilled Oils, Fuels, and Petroleum Products: 1. Composition and Properties of Selected Oils. National Exposure Research Laboratory, Office of Research and Development, United States Environmental Protection Agency, Research Triangle Park, North Carolina.

Weber, D.D., Maynard, D.J., Gronlund, W.D. and V. Konchin. 1981. Avoidance reactions of migrating adult salmon to petroleum hydrocarbons. *Canadian Journal of Fisheries and Aquatic Sciences* 38: 779-781.

Wernick, B., deBruyn, A.H., Patterson, L., and P. Chapman. 2009. Effects of an oil spill on the regrowth of emergent vegetation in a Northern Alberta lake. *Archives of Environmental Contamination and Toxicology* 57: 697-706.

Whitehead, A., Dubansky, B., Bodinier, C., Garcia, T.I., Miles, S., Pilley, C., Raghunathan, V., Roach, J.L., Walker, N., Walter, R.B., Rice, C.D. and F. Galvez. 2012. Genomic and physiological footprint of the *Deepwater Horizon* oil spill on resident marsh fishes. *Proceedings of the National Academy of Sciences* 109(50): 20298-20302.

Wiese, F.K. and P.C. Ryan. 2003. The extent of chronic marine oil pollution in southeastern Newfoundland waters assessed through beached bird surveys 1984 -1999. *Marine Pollution Bulletin* 46(9): 1090-1101.

Wikelski, M., Wong, V., Chevalier, B., Rattenborg, N. and H.L. Snell. 2002. Galapagos Islands: Marine iguanas die from trace oil pollution. *Nature* 417(6889): 607-608.

Williams, R., Gero, S., Bejder, L., Calambokidis, J., Kraus, S.D., Lusseau, D., Read, A.J., and J. Robbins. 2011. Underestimating the damage: interpreting cetacean carcass recoveries in the context of the Deepwater Horizon/BP incident. *Conservation Letters* 4: 228-233.

Wise, J.P., Jr., Wise, J.T.F., Wise, C.F., Wise, S.S., Gianios, C., Jr., Xie, H., Thompson, W.D., Perkins, C., Falank, C., and J.P. Wise, Sr. 2014. Concentrations of the genotoxic metals, chromium and nickel, in whales, tar balls, oil slicks, and released oil from the Gulf of Mexico in the immediate aftermath of the Deepwater Horizon oil crisis: Is genotoxic metal exposure part of the Deepwater Horizon legacy? *Environmental Science & Technology* 48: 2997-3006.

Wu, D., Wang, Z., Hollebone, B., McIntosh, S., King, T., and P.V. Hodson. 2012. Comparative toxicity of four chemically -dispersed and undispersed crude oils to rainbow trout embryos. *Environmental Toxicology and Chemistry* 31: 754-765.

Yang, C., Wang, Z., Yang, Z., Holleb one, B., Brown, C.E., Landriault, M. and B. Fieldhouse. 2011. Chemical fingerprints of Alberta Oil sands and related petroleum products. *Environmental Forensics* 12:173-188.

Zhang, J.F., Wang, X.R., Guo, H.Y., Wu, J.C., and Y.Q. Xue. 2004. Effects of water -soluble fractions of diesel oil on the antioxidant defenses of the goldfish, *Carassius auratus* . *Ecotoxicology and Environmental Safety* 58(1): 110-116.

ED_001324_00007647-00421

Chapter 5

Aeppli, C., Carmichael, C.A., Nelson, R.K., Lemkau, K.L., Graham, W.M., Redmond, M.C., Valentine, D.L., and C.M. Reddy. 2012. Oil weathering after the Deepwater Horizon disaster led to the formation of oxygenated residues. *Environmental Science & Technology* 46(16): 8799-8807.

Ajijolaiya, L. O., Hill, P.S., Khelifia, A., Islam, R.M., and K. Lee. 2006. Laboratory investigation of the effects of mineral size and concentration on the formation of oil–mineral aggregates. *Marine pollution* bulletin 52(8): 920-927.

Allen, A.A. 1983. Oil Spill Response in the Arctic. Part 2: Field Demonstrations in Broken Ice. Prepared for Shell Oil Company, Sohio Alaska Petroleum Company, Exxon Company, U.S.A., Amoco Production, Archorage, AK.

ASCE Task Committee on Modeling of Oil Spills of the Water Resources Engineering Division. 1996. State-of-the-art review of modeling transport and fate of oil spills. *Journal of Hydraulic Engineering* 122(11): 594-609.

Atlas, R. M. and R. Bartha. 1972. Degradation and mineralization of petroleum in sea water: limitation by nitrogen and phosphorus. *Biotechnology and Bioengineering* 14: 309-318.

Atlas, R. and T. Hazen. 2011. Oil biodegradation and bioremediation: A tale of the two worst spills in U.S. history. *Environmental Science & Technology* 45: 6709–6715.

Barker, C. 2011. A statistical outlook for the Deepwater Horizon oil spill. *Geophysical Monograph Series* 195: 237-244.

Blokker, P. 1964. Spreading and evaporation of petroleum products on water. In: *Proceedings of the Fourth International Harbour Conference, Antwerp, Belgium*. pp. 911-919.

Bobra, M. 1992. A Study of the Evaporation Of Petroleum Oils. Environment Canada, Environmental Protection Directorate, River Road Environmental Technology Centre.

Bobra, M and Fingas, M. 1986. The behaviour and fate of arctic oil spills. W*ater Science & Technology* 18:13–23.

Borden, R. C. and P. B. Bedient. 1986. Transport of dissolved hydrocarbons influenced by oxygen-limited biodegradation. 1. Theoretical development. *Water Resources Research* 22(13): 1973-1982.

Boufadel, M. C. and A. M. Bobo. 2011. Feasibility of high pressure injection of chemicals into the subsurface for the bioremediation of the Exxon Valdez oil. *Ground Water Monitoring and Remediation* 31(1): 59-67.

Boufadel, M. C., Suidan, M.T. and A.D. Venosa. 1999. A numerical model for density-and-viscosity-dependent flows in two- dimensional variably-saturated porous media. *Journal of Contaminant Hydrology* 37: 1-20.

Boufadel, M. C., Bechtel, R.D., and J. Weaver. 2006. The movement of oil under non-breaking waves. *Marine pollution Bulletin* 52(9): 1056-1065.

Boufadel, M. C., Du, K., Kaku, V. and J. Weaver. 2007. Lagrangian simulation of oil droplets transport due to regular waves. *Environmental Modelling & Software* 22(7): 978-986.

Boufadel, M.C., Sharifi, Y., Van Aken, B., Wrenn, B.A. and K. Lee. 2010. Nutrient and oxygen concentrations within the sediments of an Alaskan beach polluted with the *Exxon Valdez* oil spill. *Environmental Science & Technology* 44: 7418-7424.

Boufadel, M. C., Bobo, A.M. and Y. Xia. 2011. Feasibility of Deep Nutrients Delivery into a Prince William Sound Beach for the Bioremediation of the Exxon Valdez Oil Spill. *Ground Water Monitoring and Remediation* 31(2): 80-91.

Boufadel, M.C., Abdollahi-Nasab, A., Geng, X., Galt, J. and J. Torlapati. 2014. Simulation of the landfall of the Deepwater Horizon oil on the shorelines of the Gulf of Mexico. *Environmental Science & Technology* 48: 9496-9505.

Bragg, J.R., Prince, R.C., Harner, E.J. and R.M. Atlas. 1994. Effectiveness of bioremediation for the *Exxon Valdez* oil spill. *Nature* 368: 413-418.

Brandvik, P. J, Johansen, O., Leirvik, F., Farooq, U. and P. S. Daling. 2013. Droplet breakup in sub-surface oil releases—Part 1: Experimental study of droplet breakup and effectiveness of dispersant injection. *Marine Pollution Bulletin* 73: 319-326.

Brost, E. J. and G.E. DuVall. 2000. Non-aqueous phase liquid (NAPL) mobility limits in soil. *API Soil & Groundwater Research Bulletin* No.9.

Brustaert, W. 1982. Evaporation into the Atmosphere: Theory, History and Application. D. Reidel Publising Company, Boston.

Buffington, J. M. and D. Tonina. 2009. Hyporheic exchange in mountain rivers II: Effects of channel morphology on mechanics, scales, and rates of exchange. *Geography Compass* 3(3): 1038-1062.

Camilli, R., Di Iorio, D., Bowen, A., Reddy, C.M., Techet, A.H., Yoerger, D.R., Whitcomb, L.L., Seewald, J.S., Sylva, S.P. and J. Fenwick. 2012. Acoustic measurement of the Deepwater Horizon Macondo well flow rate. *Proceedings of the National Academy of Sciences USA* 109(50): 20235-20239.

Camus, L., Olsen, G.H. and M. Carroll. 2008. Arctic Ecotoxicological Studies Review. Akvaplanniva AS Report 3975. Tromsø, Norway.

Carls, M.G., Thomas, R.E., Lilly, M.R. and S.D. Rice. 2003. Mechanism for transport of oil-contaminated groundwater into pink salmon redds. Marine Ecology Progress Series 248: 245–255.

Chapelle, F. 2001. Ground-Water Microbiology and Geochemistry, John Wiley & Sons.

Clark, M. M. 1996. Transport Modeling for Environmental Engineers and Scientists. John Wiley and Sons Inc., New York.

Clement, T. 1997. A Modular Computer Code for Simulating Reactive Multispecies Transport in 3–Dimensional Groundwater Aquifers. Pacific Northwest National Laboratory, Richland, WA. http://bioprocess. pnl. gov/rt3d. htm.

Cozzarelli, I. M., Bekins, B.A., Eganhouse, R.P., Warren, E. and H.I. Essaid. 2010. In situ measurements of volatile aromatic hydrocarbon biodegradation rates in groundwater. *Journal of Contaminant Hydrology* 111(1-4): 48-64.

D'Auria, M., Emanuele, L., Racioppi, R. and V. Velluzzi V. 2009. Photochemical degradation of crude oil: Comparison between direct irradiation, photocatalysis, and photocatalysis on zeolite. *Journal of hazardous materials* 164(1): 32-38.

Dean, R. G. and R. A. Dalrymple. 1984. Water Wave Mechanics for Engineers and Scientists. Prentice-Hall,Inc., New Jersey.

Delvigne, G. A. L. 1994. Natural and chemical dispersion of oil. *Journal of Advanced Marine Technology* 11: 23-63.

Delvigne, G. A. L. and C. E. Sweeney. 1988. Natural dispersion of oil. *Oil & Chemical Pollution* 4: 281-310.

Dickins, D., Brandvik, P.J., Bradford, J., Faksness, L-G., Liberty, L. and R. Daniloff. 2008. Svalbard 2006 experimental oil spill under ice: Remote sensing, oil weathering under Arctic conditions and assessment of oil removal by in-situ burning. In: *Proceedings of the 2008 International Oil Spill Conference*. American Petroleum Institute. pp. 681-688.

Dickins, D., Hearon, G., Morris, K., Ambrosius, K. and W. Horowitz. 2011. Mapping sea ice overflood using remote sensing: Alaskan Beaufort Sea. *Cold Regions Science and Technology* 65(3): 275-285.

El-Kadi, A. I. 2001. Modeling hydrocarbon biodegradation in tidal aquifers with water-saturation and heat inhibition effects. *Journal of Contaminant Hydrology* 51(1-2): 97-125.

Elliot, A. J., Hurford, N. and C.J. Penn. 1986. Shear diffusion and the spreading of oil slicks. *Marine pollution Bulletin* 17(7): 308-313.

Elliott, A. H. and N. H. Brooks. 1997. Transfer of nonsorbing solutes to a streambed with bed forms: Laboratory experiments. *Water Resources Research* 33(1): 137-151.

Essaid, H. I., Bekins, B.A., Godsy, E.M., Warren, E., Baedecker, M.J. and Cozzarelli, I.M. 1995. Simulation of aerobic and anaerobic biodegradation processes at a crude oil spill site. *Water Resources Research* 31(12): 3309-3327.

Essaid, H. I., Cozzarelli, I. M., Eganhouse, R.P., Herkelrath, W.N., Bekins, B.A., and Delin, G.N. 2003. Inverse modeling of BTEX dissolution and biodegradation at the Bemidji, MN crude-oil spill site. *Journal of Contaminant Hydrology* 67(1): 269-299.

Etkin, D. S., Michel, J., French McCay, D., Boufadel, M. and H. Li. 2008. Integrating state-of-the-art shoreline interaction knowledge into spill modeling. In: *Proceedings of the 2008 International Oil Spill Conference*. American Petroleum Institute.

Farmer, D. M. and M. Li. 1994. Oil dispersion by turbulence and coherent circulations. *Ocean Engineering* 21(6): 575-586.

Fay, J. A. 1971. The Spread of Oil Slicks on a Calm Sea. Dissertation/Thesis. Cambridge, MA, Massachusetts Institute of Technology.

Fingas, M. 2011. Evaporation modeling. Chapter 9. In: Fingas, M.F. (ed.). *Oil Spill Science and Technology: Prevention, Response, and Cleanup.* Elsevier/Gulf Professional Publication. pp. 201-242.

Fingas, M. 2013. Modeling oil and petroleum evaporation. *Journal of Petroleum Science Research* 2(3): 104-115.

Fingas, M. and B. Fieldhouse. 2004. Formation of water-in-oil emulsions and application to oil spill modelling. *Journal of Hazardous Materials* 107(1): 37-50.

Fingas, M. and B. Hollebone. 2003. Review of behaviour of oil in freezing environments. *Marine Pollution Bulletin* 47(9): 333-340.

Fingas, M., Fieldhouse, B. and J. Mullin. 1996. Studies of water-in-oil emulsions: The role of asphaltenes and resins. In: *Proceedings of the Nineteenth Arctic and Marine Oilspill Program (AMOP) Technical Seminar*, Calgary, Alberta, Canada. Environment Canada. p. 73-88.

Fingas, M., Fieldhouse, B., Lane, J., and J. Mullin. 2000. Studies of water-in-oil emulsions: Long-term stability, oil properties, and emulsions formed at sea. In: *Proceedings of the Twenty-Third Arctic and Marine Oil Spill Program (AMOP) Technical Seminar*, Vancouver, British Columbia. Environment Canada. p. 145-160.

Fingas, M. F., Fieldhouse,B., Lambert, P., Wang, Z., Noonan, J. Lane, J. and J. Mullin. 2002. Water-in-oil emulsions formed at sea, in test tanks, and in the laboratory. Environment Canada, Ottawa, ON. Environment Canada Manuscript Report EE-169.

Fieldhouse, B., Lambert, P., Wang, Z., Noonan, J., Lane, J. and J.V. Mullin. 2002. Water-in-oil Emulsions Formed at Sea, in Test Tanks, and in the Laboratory. Environment Canada Manuscript Report EE-169.Environment Canada, Ottawa, Ontario.

Fischer, H. B., List, J.E., Koh, C.R., Imberger, J. and N.H. Brooks. 1979. Mixing in Inland and Coastal Waters. Academic Press, New York.

Fitzpatrick, F.A., Boufadel, M.C., Johnson, R., Lee, K., Graan, T.P., Bejarano, A.C., Zhu, Z., Waterman, D., Capone, D.M., Hayter, E., Hamilton, S.K., Dekker, T., Garcia, M.H. and J.S. Hassan. 2015. Oil-Particle Interactions and Submergence from Crude Oil Spills in Marine and Freshwater Environments – Review of the Science and Future Science Needs. U.S. Geological Survey Open File Report 2015-1076. http://dx.doi.org/10.3133/ofr20151076.

Frelichowska, J., Bolzinger, M.A., and Y. Chevalier. 2010. Effects of solid particle content on properties of o/w Pickering emulsions. *Journal of Colloid And Interface Science* 351(2): 348-356.

French-McCay, D. P. 2004. Oil spill impact modeling: Development and validation. *Environmental Toxicology and Chemistry* 23(10): 2441-2456.

Garrett, R. M., Pickering, I.J., Haith, C.E. and R. C. Prince. 1998. Photooxidation of crude oils. *Environmental Science & Technology* 32: 3719-3723.

Geng, X., Boufadel, M.C. and B. Wrenn. 2013. Mathematical modeling of the biodegradation of residual hydrocarbon in a variably-saturated sand column. *Biodegradation* 24(2): 153-163.

Geng, X., Boufadel, M.C., Personna, Y.R., Lee, K., Tsao, D. and E.D. Demicco. 2014. BioB: a mathematical model for the biodegradation of low solubility hydrocarbons. *Marine Pollution Bulletin* 83(1): 138-147.

Geng, X., Boufadel, M.C., Lee, K., Abrams, S. and M. Suidan. 2015. Biodegradation of subsurface oil in a tidally influenced sand beach: Impact of hydraulics and interaction with pore water chemistry. *Water Resources Research* 51: 3193–3218.

Gong, Y., Zhao, X., Cai, Z., O'Reilly, S.E., Hao, X., and D. Zhao. 2014. A review of oil, dispersed oil and sediment interactions in the aquatic environment: Influence on the fate, transport and remediation of oil spills. *Marine Pollution Bulletin* 79: 16-33.

Guo, Q., Li, *H.,* Boufadel,*M.C. and* Y. Sharifi. 2010. Hydrodynamics in a gravel beach and its impact on the Exxon Valdez oil. *Journal of Geophysical Research* 115: *C12077.*

Hamoda, M., Hamam, S.E.M. and S.B. Shaban. 1989. Volatilization of crude oil from saline water. *Oil and Chemical Pollution* 5(5): 321-331.

Harvey, J. W. and C. C. Fuller. 1998. Effect of enhanced manganese oxidation in the hyporheic zone on basin-scale geochemical mass balance. *Water Resources Research* 34(4): 623-636.

Hodson, P. V., Collier, T. K.and J. D. Martin. 2011. Toxicity to Fish -Potential Effects on an Oil Spill into the Kitimat River from a Northern Gateway Pipeline Rupture. https://docs.neb-one.gc.ca/ll-eng/llisapi.dll/fetch/2000/90464/90552/384192/620327/624910/693017/776343/D80-27-05_-

ED_001324_00007647-00424

_Haisla_Nation_-_Hodson_Collier_Martin-Toxicity_of_Oil_to_Fish_-_A2K3D7.pdf?nodeid=776350&vernum=-2
Hoult, D. P. 1972. Oil spreading on the sea. *Annual Review of Fluid Mechanics* 4: 341-368.
Johansen, Ø. 2000. Deepblow-A lagrangian plume model for deep water blowouts. *Spill Science and Technology Bulletin* **6**(2): 103-111.
Johansen, Ø., Rye, H., Melbye, A.G., Jensen, H.V., Serigstad, B. and T. Knutsen. 2001. Deep Spill JIP - Experimental Discharges of Gas and Oil at Helland Hansen - June 2000. Technical Report. Prepared for DeepSpill JIP. SINTEF, Trondheim, Norway.
Johansen, Ø., Brandvik, P.J. and U.Forooq. 2013. Droplet breakup in subsea oil releases – Part 2: Predictions of droplet size distributions with and without injection of chemical dispersants. *Marine Pollution Bulletin* 73: 327-335.
Jokuty, P., Fingas, M., Whiticar, S. and B. Fieldhouse. 1995. A Study of Viscosity and Interfacial Tension of Oils and Emulsions. Environmental Protection Service, Environment Canada, Ottawa, ON.
Jones, J. B. and P. J. Mulholland. 1999. Streams and Ground Waters. Academic Press.
Kazunga, C. and M. D. Aitken. 2000. Products from the incomplete metabolism of pyrene by polycyclic aromatic hydrocarbon-degrading bacteria. *Applied and Environmental Microbiology* 66(5): 1917-1922.
Kindred, J. S. and M. A. Celia. 1989. Contaminant transport and biodegradation: 2. Conceptual model and test simulations. *Water Resources Research* 25(6): 1149-1159.
King, T. L., Clyburne, J.A., Lee, K. and B.J. Robinson. 2013. Interfacial film formation: Influence on oil spreading rates in lab basin tests and dispersant effectiveness testing in a wave tank. *Marine Pollution Bulletin* 71: 83-91.
Kujawinski, E.B., Kido Soule, M.C., Valentine, D.L., Boysen, A.K., Longnecker, K. and M.C. Redmond. 2011. Fate of dispersants associated with the Deepwater Horizon spill. *Environmental Science and Technology* 45: 1298-1306.
Langmuir, I. 1938. Surface motion of water induced by wind. *Science* 87(2250): 119-123.
Le Floch, S., Guyomarch, J., Merlin, F-X., Stoffyn-Egli, P. Dixon, J. and K. Lee.2002. The influence of salinity on oil–mineral aggregate formation. *Spill Science & Technology Bulletin* 8(1): 65-71.
Lee, K. 2002. Oil–particle interactions in aquatic environments: Influence on the transport, fate, effect and remediation of oil spills. *Spill Science & Technology Bulletin* 8(1): 3-8.
Lee, K. and P. Stoffyn-Egli. 2001. Characterization of oil mineral aggregates. In: *Proceedings of the 2001 International Oil Spill Conference*, Tampa, Florida, March 26-29, 2001. pp. 991-996.
Lee, K., Boudreau, M., Bugden, J., Burridge, L., Cobanli, S.E., Courtenay, S., Grenon, S., Hollebone, B., Kepkay, P., Li, Z., Lyons, M., Niu, H., King, T.L., MacDonald, S., McIntyre, E.C., Robinson, B., Ryan, S.S. and G. Wohlgeschaffen. 2011. State of Knowledge Review of Fate and Effect of Oil in the Arctic Marine Environment. A report prepared for the National Energy Board of Canada Arctic Roundtable, Fisheries and Oceans Canada. 01/2011; National Energy Board of Canada. https://www.neb-one.gc.ca/ll-eng/livelink.exe/fetch/2000/90463/621169/700096/704342/A2A8R2_-_NEB_State_of_Knowledge_Review_of_Fate_and_Effect_of_Oil_DFO_COOGER.pdf?nodeid=704343&vernum=0.
Lehr, W., Fraga, R. J., Belen, M. S. and H.M. Cekirge. 1984. A new technique to estimate initial spill size using a modified Fay-type spreading formula. *Marine Pollution Bulletin* 15(9): 326-329.
Leibovich, S. and D. Ulrich. 1972. A note on the growth of small-scale Langmuir circulations. *Journal of Geophysical Research* 77(9): 1683-1688.
Li, H. and M. C. Boufadel. 2010. Long-term persistence of oil from the Exxon Valdez spill in two-layer beaches. *Nature Geoscience* 3(2): 96-99.
Li, H., Zhao, Q., Boufadel, M.C. and A.D. Venosa.2007. A universal nutrient application strategy for the bioremediation of oil-polluted beaches. *Marine Pollution Bulletin* 54(8): 1146-1161.
Mackay, D. and P. L. Leinonen. 1977. Mathematical Model of the Behavior of Oil Spills on Water with NaturalaAnd Chemical Dispersion. Environment Canada, Ottawa.
Mackay, D. and W. Zagorski. 1982. Studies of Water-in-Oil Emulsions. Report prepared for Environment Canada, Environmental Protection Service, Environmental Impact Control Directorate, Environmental Emergency Branch, Research And Development Division.
Mackay, D., Buist, I., Mascarenhas, R. and S. Paterson. 1980. Oil spill Processes and Models. Environment Canada Report EE8. Ottawa, ON.

Maki, H., Sasaki, T. and S. Harayama. 2001. Photo-oxidation of biodegraded crude oil and toxicity of the photo-oxidized products. *Chemosphere* 44(5): 1145-1151.

Molz, F. J., Widdowson, M.A. and L. D. Benefield. 1986. Simulation of microbial-growth dynamics coupled to nutrient and oxygen-transport in porous-media. *Water Resources Research* 22(8): 1207-1216.

Mooney, M. 1951. The viscosity of a concentrated suspension of spherical particles. *Journal of Colloid Science* 6(2): 162-170.

Ng, C. G.-H., Bekins, B.A., Cozzarelli, I.M., Baedecker, M.J., Bennett, P.C. and R.T. Amos. 2014. A mass balance approach to investigating geochemical controls on secondary water quality impacts at a crude oil spill site near Bemidji, MN. *Journal of Contaminant Hydrology* 164: 1-15.

Nicol, J. P., Wise, W.R., Molz, F.J. and L.D. Benefield. 1994. Modeling biodegradation of residual petroleum in a saturated porous column. *Water Resources Research* 30(12): 3313-3325.

Omotoso, O.E., Munoz, V.A. and R.J. Mikula. 2002. Mechanisms of crude oil–mineral interactions. *Spill Science & Technology Bulletin* 8(1): 45-54

OSAT (Operational Science Advisory Team). 2011. Summary Report for Fate and Effects of Remnant Oil in the Beach Environment, Prepared for U.S. Coast Guard Federal On-Scene Coordinator Deepwater Horizon MC252.

Owens, E. H. and K. Lee. 2003. Interaction of oil and mineral fines on shorelines: review and assessment. *Marine Pollution Bulletin* 47(9-12): 397-405.

Prince, R. C. 1993. Petroleum spill bioremediation in marine environments. *Critical Reviews in Microbiology* 19: 217-242.

Prince, R. C., Clark, J.R., Lindstrom, J.E., Butler, E.L., Brown, E.J., Winter, G., Grossman, M.J., Parrish, P.R., Bare, R.E., Braddock, J.F., Steinhauer, W.G., Douglas, G.S., Kennedy, J.M. and P.J. Barter. 1994. Bioremediation of the Exxon Valdez oil spill: monitoring safety and efficacy. In: Hinchee, R. E., Miller, R.N. and R. E. Hoeppel (eds.) *Hydrocarbon Bioremediation*. Lewis Publishers, Boca Raton, FL. pp. 107-124.

Prince, R.C., Garrett, R. M., Bare, R.E., Grossman, M.J., Townsend, T., Suflita, J.M., Lee, K., Owens, E.H., Sergy, G.A., Braddock, J.F., Lindstrom, J.E. and R. R. Lessard. 2003. The roles of photooxidation and biodegradation in long-term weathering of crude and heavy fuel oils. *Spill Science and Technology Bulletin* 8: 145-156.

Pritchard, P.H., Mueller, J.G., Rogers, J.C., Kremer, F.V. and J.A. Glaser. 1992. Oil spill bioremediation: experiences, lessons and results from the *Exxon Valdex* oil spill in Alaska. *Biodegradation* 3: 315-335.

Radović, J. R., Aeppli, C., Nelson, R. K., Jimenez, N., Reddy, C. M., Bayona, J. M., and J. Albaigés. 2014. Assessment of photochemical processes in marine oil spill fingerprinting. *Marine Pollution Bulletin* 79(1): 268-277.

Rittmann, B. E. and P. L. McCarty. 2001. Environmental Biotechnology: Principles and Applications. McGraw-Hill, Boston.

Rønn, R., McCaig, A. E., Griffiths, B. S.and J. I. Prosser. 2002. Impact of protozoan grazing on bacterial community structure in soil microcosms. *Applied and Environmental Microbiology* 68(12): 6094-6105.

Ross, S. and I. Buist.1995. Preliminary laboratory study to determine the effect of emulsification on oil spill evaporation. In: *Proceedings of the Eighteenth Arctic and Marine Oilspill Program (AMOP) Technical Seminar*, Edmonton, Alberta. Environment Canada.

Ross, S. and D. Dickins. 1987. Field Research Spills to Investigate the Physical and Chemical Fate of Oil in Pack Ice. Environmental Studies Revolving Fund Report No. 062. Natural Resources Canada, Ottawa, ON.

Rutherford, J. C. 1994. River Mixing. John Wiley and Sons.

Ryan, R. J. and M. C. Boufadel. 2006. "Influence of streambed hydraulic conductivity on solute exchange with the hyporheic zone." Environmental Geology **51**(2): 203-210.

Ryan, R. J. and M. C. Boufadel. 2007a. Evaluation of streambed hydraulic conductivity heterogeneity in an urban watershed. *Stochastic Environmental Research and Risk Assessment* 21(4): 309-316.

Ryan, R. J. and M. C. Boufadel. 2007b. Lateral and longitudinal variation of hyporheic exchange in a piedmont stream pool. *Environmental Science & Technology* 41(12): 4221-4226.

Ryerson, T. B., Camilli, R., Kessler, J.D., Kujawinski, E.B., Reddy, C.M., Valentine, D.L., Atlas, E., Blake, D.R., de Gouw, J. and S. Meinardi. 2012. Chemical data quantify Deepwater Horizon

ED_001324_00007647-00426

hydrocarbon flow rate and environmental distribution. *Proceedings of the National Academy of Sciences* 109(50): 20246-20253.

Schirmer, M., Molson, J. W., Frind, E.O. and J. F. Barker. 2000. Biodegradation modelling of a dissolved gasoline plume applying independent laboratory and field parameters. *Journal of Contaminant Hydrology* 46(3-4): 339-374.

Short, J. 2015. Fate and Effect of Oil Spills from the Trans Mountain Expansion Project in Burrard Inlet and the Fraser River Estuary A report Prepared for Tsleil-Waututh Nation, City of Vancouver, and Living Oceans Society.

Stiver, W. and D. Mackay. 1984. Evaporation rate of spills of hydrocarbons and petroleum mixtures. *Environmental Science & Technology* 18(11): 834-840.

Stoffyn-Egli, P. and K. Lee. 2002. Formation and characterization of oil–mineral aggregates. *Spill Science & Technology Bulletin* 8(1): 31-44.

Sun, J., Khelifa, A., Zheng, X., Wang, Z., So, L.L., Wong, S., Yang, C. and B. Fieldhouse. 2010. A laboratory study on the kinetics of the formation of oil-suspended particulate matter aggregates using the NIST-1941b sediment. *Marine Pollution Bulletin* 60(10): 1701-1707.

Sutton, O. G. 1934. Wind structure and evaporation in a turbulent atmosphere. In: *Proceedings of the Royal Society of London. Series A, Containing Papers of a Mathematical and Physical Character.* pp. 701-722.

Tchobanoglous, G., Burton, F. and H. D. Stensel. 2002. Wastewater Engineering: Treatment and Reuse. Metcalf and Eddy, McGraw-Hill.

Tkalin, A. 1986. Evaporation of petroleum hydrocarbons from films on a smooth sea surface. *Oceanology ONLGAE* 26: 473-474.

Tonina, D. and J.M. Buffington. 2007. Hyporheic exchange in gravel bed rivers with pool-riffle morphology: Laboratory experiments and three-dimensional modeling. *Water Resources Research* 43(W01421).

Tonina, D. and J.M. Buffington. 2009. Hyporheic exchange in mountain rivers I: Mechanics and environmental effects. *Geography Compass* 3(3): 1063-1086.

Torlapati, J. and M. C. Boufadel. 2014. Evaluation of the biodegradation of Alaska North Slope oil in microcosms using the biodegradation model BIOB. *Frontiers in Aquatic Microbiology* 5(212): 1-15.

Tsouris, C. and L. Tavlarides. 1994. Breakage and coalescence models for drops in turbulent dispersions. *AIChE Journal* 40(3): 395-406.

Venkatesh, S., El-Tahan, H., Comfort, G. and R. Abdelnour. 1990. Modeling the behavior of oil spills in Ice-infested waters. *Atmosphere-Ocean* 28: 303-329.

Venosa, A.D and E.L. Holder. 2007. Biodegradability of dispersed crude oil and two different temperatures. *Marine Pollution Bulletin* 54(5): 545-553.

Venosa, A.D., Suidan, M.T., Wrenn, B.A., Strohmeier, K.L., Haines, J.R., Eberhart, B.L., King, D. and E. Holder. 1996. Bioremediation of an experimental oil spill on the shoreline of Delaware Bay. *Environmental Science and Technology* 30(5): 1764-1775.

Venosa, A. D., Campo, P. and M. T. Suidan. 2010. Biodegradability of lingering crude oil 19 years after the Exxon Valdez oil spill. *Environmental Science & Technology* 44(19): 7613-7621.

Vilcáez, J., Li, L. and S. S. Hubbard. 2013. A new model for the biodegradation kinetics of oil droplets: application to the Deepwater Horizon oil spill in the Gulf of Mexico. *Geochemical Transactions* 14(4).

Viveros, D., Jones, R. and M. Boufadel. 2015. The biodegradation model of the NOAA oil spill software ADIOS3. In: *Proceedings of the Thirty-Eighth Arctic and Marine Oilspill Program (AMOP)*, Vancouver, Canada, June 2-5.

Wang, W., Zheng, Y. and K. Lee. 2013. Chemical dispersion of oil with mineral fines in a low temperature environment. *Marine Pollution Bulletin* 72: 205-212

Weise, W. and G. Rheinheimer. 1977. Scanning electron microscopy and epifluorescence investigation of bacterial colonization of marine sand sediments. *Microbial Ecology* 4(3): 175-188.

Widdowson, M., Waddill, D., Brauner, J., Chapelle, F. and P. Bradley. 2002. SEAM3D: A Numerical Model for Three-Dimensional Solute Transport Coupled to Sequential Electron Acceptor-Based Biological Reactions in Groundwater. Technical Report 24061. The Via Department of Civil and Environmental Engineering, Virginia Polytechnic Institute and State University, Blacksburg, Virginia.

Wrenn, B.A., Suidan, M.T., Strohmeier, K.L., Eberhart, B.L., Wilson, G.J. and A.D. Venosa. 1997. Nutrient transport during bioremediation of contaminated beaches: Evaluation with lithium as a conservative tracer. *Water Research* 31: 515-524.

Xia, Y., Li, H., Boufadel, M.C. and Y. Sharifi. 2010. Hydrodynamic factors affecting the persistence of the Exxon Valdez oil in a shallow bedrock beach. *Water Resources Research* 46: W10528-W10528.

Xie, H., Yapa, P. D. and K. Nakata. 2007. Modeling emulsification after an oil spill in the sea. *Journal of Marine Systems* 68(3): 489-506.

Yang, W. C. and H. Wang. 1977. Modeling of oil evaporation in aqueous environment. *Water Research* 11(10): 879-887.

Yang, Z., Hollebone, B. P., Wang, Z., Yang, C. Brown, C., Zhang, G., Landriault, M. and X. Ruan. 2015. A preliminary study for the photolysis behavior of biodiesel and its blends with petroleum oil in simulated freshwater. *Fuel* 139: 248-256.

Yassine, M. H., Suidan, M. T. and A. D. Venosa. 2013. Microbial kinetic model for the degradation of poorly soluble organic materials. *Water Research* 47(4): 1585-1595.

Zeidan, E., Zahariev, K., Li, M. and C. Garrett. 1997. The breakup of oil spills in the marine environment. In: *Proceedings of 20th Arctic Marine Oilspill Program (AMOP) Technical Seminar,* Vancouver, Canada. Environment Canada.

Zhao, L., Boufadel, M.C., Socolofsky, S.A., Adams, E., King, T. and K. Lee. 2014a. Evolution of droplets in subsea oil and gas blowouts: Development and validation of the numerical model VDROP. *Journal of Marine Pollution Bulletin* 83: 58-69.

Zhao, L., Torlapati, J., Boufadel, M.C., King, T., Robinson, B. and K. Lee. 2014b. VDROP: A comprehensive model for droplet formation of oils and gases in liquids-Incorporation of the interfacial tension and droplet viscosity. *Chemical Engineering Journal* 253: 93-106.

Zhao, L., Boufadel, M. C., Adams, E.E., Socolofsky, S., King, T. and K. Lee. 2015. Simulation of scenarios of oil droplet formation from the Deepwater Horizon blowout. *Marine Pollution Bulletin*. Accepted.

Zheng, L., Yapa, P. D. and F. Chen. 2003. A model for simulating deepwater oil and gas blowouts-Part I: Theory and model formulation. *Journal of Hydraulic Research* **41**(4): 339-351.

Zhu, X., Venosa, A.D., Suidan M.T. and K. Lee. 2001. Guidelines for the Bioremediation of Marine Shorelines and Freshwater Wetlands. U.S. Environmental Protection Agency.

Chapter 6

API. 2015. Field operations guide for in-situ burning of on-water oil spills, *API Technical Report 1252, 1st Edition.* American Petroleum Institute, Washington, DC.

Andrade, F., Lutz, G., Lyra, A. P., Ranieri, A. and B. Lemos. 2013. Oil Spill Response Equipment and Techniques for Waxy Oils in Brazil. In: *Proceedings of the SPE Latin-American and Caribbean Heath, Safety, Environment and Social Responsibility Conference*. Society of Petroleum Engineers.

Arctic Oil Spill Response Technology Joint Industry Programme. 2013. Dispersant Testing Under Realistic Conditions. Final Report 2.1. Report from Joint Industry Programme (JIP) to identify and summarise the state-of-the-art on research conducted to date on the effectiveness of dispersant and mineral fines in ice. http://www.arcticresponsetechnology.org/wp-content/uploads/2013/10/Report%202.1%20-%20DISPERSANT%20TESTING%20UNDER%20REALISTIC%20CONDITIONS.pdf

Atlas, R. and T. Hazen. 2011. Oil biodegradation and bioremediation: A tale of the two worst spills in U.S. history. *Environmental Science & Technology* 45: 6709–6715.

Aurand, D. and G. Coelho. 2006a. Ecological Risk Assessment Workshop: Environmental Tradeoffs Associated with Oil Spill Response Technologies. Cape Flattery, Washington. A report to Regional Response Team X. Ecosystem Management & Associates, Inc., Lusby, MD. Technical Report 05-01.

Aurand, D. and G. Coelho. 2006b. Ecological Risk Assessment: Consensus Workshop. Environmental Tradeoffs Associated with Oil Spill Response Technologies. A Report to the US Coast Guard,

Sector Delaware Bay. Ecosystem Management & Associates, Inc., Lusby, MD. Technical Report 06-01.

Bento, F.M., de Oliveira-Camargo, F.A., Okeke, B.C. and W.T. Frankenberger, Jr. 2005. Diversity of biosurfactant producing microorganisms isolated from soils contaminated with diesel oil. *Microbiological Research* 160 (3): 249-255.

Bragg, J.R., Prince, R.C., Harner, E.J. and R.M. Atlas. 1994. Effectiveness of bioremediation for the *Exxon Valdez* oil spill. *Nature* 368: 413-418.

Brinkman, J.S. 1998. Absorbents vs. adsorbents. *Hazmat Management* Sept. 1, 1998. http://www.hazmatmag.com/features/absorbents-vs-adsorbents/

Buist I., Potter, S., Nedwed, T. and J. Mullin. 2008. Herding agents thicken oil spills in drift ice to facilitate in situ burning: A new trick for an old dog. In: *Proceedings of the 2008 International Oil Spill Conference, Savannah*. American Petroleum Institute, Washington, DC. Pp. 673–680.

Buist I., Canevari, G. and T. Nedwed. 2010a. New herding agents for thickening oil slicks in drift ice for in situ burning. In: *Proceedings of the Thirty-third AMOP Technical Seminar on Environmental Contamination and Response*. Environment Canada, Ottawa. pp 725–742.

Buist I. , Potter, S. and S.E. Sørstrøm. 2010b. Barents Sea field test of herder to thicken oil for in situ burning. In: *Proceedings of the Thirty-third AMOP Technical Seminar on Environmental Contamination and Response*. Environment Canada, Ottawa. pp 1085–1108.

Buist, I., Potter, S. and T. Nedwed. 2011. Herding agents to thicken oil spills in drift ice for in situ burning: New developments. In: *Proceedings of the 2011 International Oil Spill Conference*. American Petroleum Institute, Washington, DC. pp. abs230.

Burns, K.A., Codi, S. and N.C. Duke. 2000. Gladstone, Australia field studies: weathering and degradation of hydrocarbons in oiled mangrove and salt marsh sediments with and without the application of an experimental bioremediation protocol. *Marine Pollution Bulletin* 41: 392-402.

Cai, Q., Zhang, B., Chen, B., Zhu, Z., Lin, W. and T. Cao. 2014a. Screening of biosurfactant producers from petroleum hydrocarbon contaminated sources in cold marine environments. *Marine Pollution Bulletin* 86(1): 402-410.

Cai, Q.H., Zhang, B.Y., Chen, B. and T. Cao. 2014b. Biodispersant produced by a Rhodococcus Erythropolis mutant as an oil spill response agent. In: *Proceedings of the International Conference on Marine and Freshwater Environments (iMFE 2014), August 6-8, St. John's, Canada*. MOS1137:1-12.

Canadian Coast Guard. 2005. Oil Spill Response Field Guide (4th Printing).

Chroma, L., Mackova, M., Kucerova, P., der Wiesche, C., Burkhard, J. and T. Macek. 2002. Enzymes in plant metabolism of PCBs and PAHs. *Acta Biotechnologica* 22 (1-2): 35–41.

Chung, S., Suidan, M.T. and A.D. Venosa, 2012. Effectiveness of plant-derived sorbents for the remediation of low-energy intertidal wetlands contaminated by oil spills. *Water, Air, and Soil Pollution* 223 (6): 3137-3144.

Clayton, J.R., Payne, J.R. and J.S. Farlow. 1993. Oil Spill Dispersants: Mechanisms of Action and Laboratory Tests. CRC Press, Inc., Boca Raton, FL.

Colten, C. E., Hay, J. and A. Giancarlo. 2012. Community resilience and oil spills in coastal Louisiana. *Ecology and Society* 17(3): 5.

Criquet, S., Joner, E., Leglize, P. and C. Leyval. 2000. Anthracene and mycorrhiza affect the activity of oxidoreductases in the roots and the rhizosphere of lucerne (*Medicago sativa* L.). *Biotechnology Letters* 22 (21): 1733–1737.

Crosby, S., Fay, R., Groark, C., Kani, A., Smith, J.R. and T. Sullivan. 2013. Transporting Alberta's Oil Sands Products: Defining the Issues and Assessing the Risks. http://www.rrt10nwac.com/Files/FactSheets/131217071129.pdf

Cunningham, S.D., Anderson, T.A., Schwab, A.P. and F.C. Hsu. 1996. Phytoremediation of soils contaminated with organic pollutants. *Advances in Agronomy* 56: 55-114.

Dahl, W.A., Lessard, R.R. and E.A. Cardello. 1997. Recent research on the application and practical effects of solidifiers. In: *Proceedings of the 1997 International Oil Spill Conference*. American Petroleum Institute, Washington, DC. Pp. 391-395.

Daling, P.S., Holumsnes, A., Rasmussen, C., Brandvik, P.J. and F. Leirvik. 2010. Development and Testing of a Containerized Dispersant Spray System for Use in Cold and Ice-Covered Areas. Oil-in-Ice JIP Report No. 13/ARCTECH P4. SINTEF Materials and Chemistry, Trondheim, Norway. http://www.sintef.no/project/JIP_Oil_In_Ice/Dokumenter/publications/JIP-rep-no-13-Development-of-spray-arm-final.pdf. (accessed June 12, 2015).

Dejhosseini, M., Aida, T., Watanabe, M., Takami, S., Hojo, D., Aoki, N., Arita, T., Kishita, A. and T. Adschiri. 2013. Catalytic cracking reaction of heavy oil in the presence of cerium oxide nanoparticles in supercritical water. *Energy & Fuels* 27(8): 4624-4631.

Delaune, R.D., Lindau, C.W. and A. Jugsujinda. 1999. Effectiveness of "Nochar" solidifier polymer in removing oil from open water in coastal wetlands. Spill *Science and Technology Bulletin* 5: 357–359.

Demirbas, A., Alidrisi, H. and M.A. Balubaid. 2015. API gravity, sulfur content, and desulfurization of crude oil. *Petroleum Science and Technology* 33(1): 93-101.

Deng, D., Prendergast, D. P., MacFarlane, J., Bagatin, R., Stellacci, F. and P.M. Gschwend. 2013. Hydrophobic meshes for oil spill recovery devices. *ACS Applied Materials & Interfaces* 5(3): 774-781.

Dickins, D. 2011. Behavior of oil spills in ice and implications for arctic spill response. In: *Arctic Technology Conference, OTC 22126, February 7-9, Texas*. Offshore Technology Conference.

Duke, N.C., Burns, K.A., Swannell, R. P. J., Dalhaus, O. and R.D. Rupp. 2000. Dispersant use and a bioremediation strategy as alternate means of reducing impacts of large oil spills on mangroves: The Gladstone field trials. *Marine Pollution Bulletin* 41: 403-412.

Doerffer, J.W. 1992. Oil Spill Response in the Marine Environment. Pergamon Press, Oxford, U.K.

Doerffer, J. W. 2013. Oil spill Response in the Marine Environment. Elsevier.

EPPR (Emergency Prevention, Preparedness and Response). 2015. Guide to Oil Spill Response in Snow and Ice Conditions. Narayana Press, Denmark.

Etkin, D.S. and P. Tebeau. 2003. Assessing progress and benefits of oil spill response technology development since Exxon Valdez. In: *Proceedings of the 2003 International Oil Spill Conference.* American Petroleum Institute, Washington, D.C.

Favas, P.J., Pratas, C.J., Varun, M., D'Souza, R. and M.S. Paul. 2014. Chapter 17. Phytoremediation of Soils Contaminated with Metals and Metalloids at Mining Areas: Potential of Native Flora. In: Soriano, M.C. (ed.) *Environmental Risk Assessment of Soil Contamination*. InTech: DOI: 10.5772/57086.

Federal Interagency Solutions Group. 2010. Oil Budget Calculator: Deepwater Horizon. A Report to the National Incident Command, NOAA, USGS, and NIST. http://www.noaanews.noaa.gov/stories2010/PDFs/OilBudgetCalc_Full_HQ-Print_111110.pdf

Ferek, R., Allen, A. and J.H. Kucklick. 1997. Air Quality Considerations Involving In-Situ Burning. Marine Preservation Association, Scottsdale, AZ.

Ferro, A. M., Kennedy, J., Doucette, W., Nelson, S., Jauregui, G., McFarland, B. and B. Bugbee. 1997. Fate of benzene in soils planted with alfalfa: uptake, volatilization, and degradation. In: Kruger, E.L., Anderson, T.A. and J.R. Coates (eds) *Phytoremediation of Soil and Water Contaminants.* American Chemical Society, Washington, DC. pp. 223-237.

Fieldhouse, B. and M. Fingas. 2009. Solidifier effectiveness: variation due to oil composition, oil thickness and temperature. In: *Proceedings of the 32nd AMOP Technical Seminar on Environmental Contamination and Response*. Environment Canada, Ottawa. 337-363.

Fingas, M. 2008. A Review of Literature Related to Oil Spill Solidifiers 1990–2008. Prince Williams Sound Regional Citizens' Advisory Council (PWSRCAC), Anchorage, AK.

Fingas, M.F. 2011. Oil Spill Science and Technology: Prevention, Response, and Cleanup. Elsevier/Gulf Professional Publication.

ED_001324_00007647-00430

Fingas, M.F. 2012. Buoys and devices for oil spill tracking. In: *Proceedings of the 35th AMOP Technical Seminar on Environmental Contamination and Response, Vancouver, Canada.* Environment Canada, Ottawa. Pp. 213-228.

Fingas, M. 2013a. Review of Solidifiers: An Update 2013. Prince Williams Sound Regional Citizens' Advisory Council (PWSRCAC). http://www.pwsrcac.org/resources/reports-documents/

Fingas, M. 2013b. Review of Oil Spill Herders. Prince Williams Sound Regional Citizens' Advisory Council (PWSRCAC). http://www.pwsrcac.org/resources/reports-documents/

Fingas, M. 2013c. Macondo well blowout mass balance: a chemical view. In: *Proceedings of the 36th AMOP Technical Seminar on Environmental Contamination and Response*, Environment Canada, Ottawa.

Fingas, M.F., Kolokowski, B. and E.J. Tennyson. 1990a. Study of oil spill dispersants effectiveness and physical studies. In: *Proceedings of the 13th Arctic and Marine Oilspill Program Technical Seminar, Edmonton, Alberta*. Environment Canada. Pp. 265-287.

Fingas, M.F., Stoodley, R. and N. Laroche. 1990b. Effectiveness testing of spill-treating agents. *Oil Chemistry and Pollution* 7: 337–348.

Fingas, M.F., Halley, G., Ackerman, F., Nelson, R., Bissonnette, M.C., Laroche, N., Wang, Z., Lambert, P., Li, K., Jokuty, P., Sergy, G., Halley, W., Latour, J., Galarneau, R., Ryan, B., Campagna, P.R., Turpin,R.D., Tennyson, E.J., Mullin, J., Hannon, L., Aurand, D. and R. Hiltabrand, 1995a. The Newfoundland Offshore Burn Experiment. In: *Proceedings of the 1995 International Oil Spill Conference.* American Petroleum Institute, Washington, D.C. pp. 123-132.

Fingas, M.F., Kyle, D.A., Larouche, N., Fieldhouse, B., Sergy, G. and G. Stoodley. 1995b. Effectiveness Testing of Oil Spill-treating Agents. ASTM Special Technical Publication 1252. pp. 286–298.

Forsyth, J.V., Tsao, Y.M. and R.D. Blem. 1995. Bioremediation: when is augmentation needed? In: Hinchee, R.E., Alleman, B.C. and J. Frederickson (eds.) *Bioaugmentation for Site Remediation.* Battelle Press, Columbus, OH. pp1-14.

Frick, C.M., Germida, J.J. and R.E. Farrell. 1999. Assessment of phytoremediation as an in-situ technique for cleaning oil-contaminated sites. In: *Proceedings of the Phytoremediation Technical Seminar.* Environment Canada, Ottawa. Pp. 105-124.

Frielo, D.A., Mylroie, J.R. and A.M. Chakrabarty. 2001. Use of genetically engineered multi-plasmid microorganisms for rapid degradation of fuel hydrocarbons. *International Biodeterioration & Biodegradation* 48: 233–242.

Garcia-Blanco, S., Venosa, A.D., Suidan, M.T., Lee, K., Cobanli, S. and J.R. Haines. 2007. Biostimulation for the treatment of an oil-contaminated coastal salt marsh. *Biodegradation* 18(1): 1-15.

Garrett W.D. and W.R. Barger. 1972. Control and Confinement of Oil Pollution on Water with Monomolecular Surface Films. Final Report to U.S. Coast Guard, Nov 1971, Project No. 724110.1/4.1 (also reprinted as U.S. Navy Naval Research Laboratory Memorandum Report 2451, June 1972, AD 744–943).

Genwest Systems, Inc. 2012. EDRC Project Final Report. GSA Contract GS-00F-0002W, BSEE Order #E12-PD-00012. Prepared for Bureau of Safety and Environmental Enforcement. http://www.genwest.com/Genwest_EDRC-Project_Final_Report.pdf.

Ghalambor, A. 1997. The Effectiveness of Solidifiers for Combating Oil Spills. Louisiana Applied and Educational Oil Spill Research and Development Program. Baton Rouge, LA. Technical Report Series, 96-006.

Gong, K., Tkalich, P. and H. Xu. 2014. The numerical investigation on oil slick behavior behind the oil boom. *Journal of Environmental Protection* 5(8): 739.

Goodbody-Gringley, G., Wetzel, D. L., Gillon, D., Pulster, E., Miller, A. and K.B. Ritchie. 2013. Toxicity of Deepwater Horizon source oil and the chemical dispersant, Corexit® 9500, to coral larvae. *PLoS One* 8(1): e45574.

Gramss, G., Voigt, K-D. and B. Kirshe, 1999. Oxidoreductase enzymes liberated by plant roots and their effects on soil humic material. *Chemosphere* 38(7): 1481-1494.

Gray, J.L., Kanagy, L.K., Furlong, E.T., McCoy, J.W. and C.J. Kanagy. 2011. Determination of the Anionic Surfactant Di(Ethylhexyl) Sodium Sulfosuccinate in Water Samples Collected from Gulf of Mexico Coastal Waters Before and After Landfall of Oil From the Deepwater Horizon Oil Spill, May to October, 2010: U.S. Geological Survey Open-File Report 2010–1318. http://pubs.usgs.gov/of/2010/1318/.

Greer, C.W., Fortin, N., Sanschagrin, S., Yergeau, E., Whyte, L.G., King, T.L. and K. Lee. 2015. Natural degradation potential for crude oil at sub-zero temperatures in the Canadian Arctic marine environment. Abstract, *38th Arctic and Marine Oilspill Program (AMOP) Technical Seminar.* Environment Canada, Ottawa.

Gülüm, M. and A. Bilgin. 2015. Density, flash point and heating value variations of corn oil biodiesel–diesel fuel blends. *Fuel Processing Technology* 134: 456-464.

Hassanshahian, M., Emtiazi, G. and S. Cappello. 2012. Isolation and characterization of crude-oil-degrading bacteria from the Persian Gulf and the Caspian Sea. *Marine Pollution Bulletin* 64(1): 7-12.

Hoffman, A. J. and P.D. Jennings. 2011. The BP oil spill as a cultural anomaly? Institutional context, conflict, and change. *Journal of Management Inquiry*: DOI: 10.1177/1056492610394940.

Hu, G., Li, J. and G. Zeng. 2013. Recent development in the treatment of oily sludge from petroleum industry: A review. *Journal of Hazardous Wastes* 261: 470-490.

IMO. 2005. Manual on Oil Pollution: Section IV – Combating Oil Spills (2nd edition). International Maritime Organization.

ITOPF (International Tanker Owners Pollution Federation Ltd.). 2014. Use of Dispersants to Treat Oil Spills. Technical Information Paper #4. London, UK. http://www.itopf.com/knowledge-resources/documents-guides/document/tip-4-use-of-dispersants-to-treat-oil-spills/.

ITOPF. 2014. Use of Sorbent Materials in Oil Spill Response. Technical Information Paper #8. London. http://www.itopf.com/knowledge-resources/documents-guides/document/tip-8-use-of-sorbent-materials-in-oil-spill-response/.

Ivanov, B., Bezgreshnov, A., Kubyshkin, N. and A. Kursheva. 2005. Spreading of oil products in sea ice and their influence on the radiation properties of the snow-ice cover. In: *Proceedings of the 18th International Port and Oceans Engineering Under Arctic Conditions.* Vol. 2. pp. 853-862.

Jacobs, T., Jacobi, M., Rogers, M., Adams, J., Coffey, J., Walker, J. and B. Johnston. 2015. Testing and evaluating low altitude unmanned aircraft system technology for maritime domain awareness and oil spill response in the arctic. *Marine Technology Society Journal* 49(2): 145-150.

Jacques, R.J.S., Okeke, B.C., Bento, F.M., Teixeira, A.S., Peralba, M.C.R. and F.A.O. Camargo. 2008. Microbial consortium bioaugmentation of polycyclic aromatic hydrocarbons in contaminated soil. *Bioresource Technology* 99: 2637-2643.

JRP (Joint Review Panel). 2013. Connections: Report of the Joint Review Panel for the Enbridge Northern Gateway Project, Vol. 1. National Energy Board, Calgary, AB.. http://gatewaypanel.review-examen.gc.ca/clf-nsi/dcmnt/rcmndtnsrprt/rcmndtnsrprtvlm1-eng.pdf.

Kleindienst, S., Paul, J. H. and S.B. Joye. 2015. Using dispersants after oil spills: impacts on the composition and activity of microbial communities. *Nature Reviews Microbiology* 13(6): 388-396.

Kok, M. V. 2011. Characterization of medium and heavy crude oils using thermal analysis techniques. *Fuel Processing Technology* 92(5): 1026-1031.

Kraus, J.J., Munir, I.Z., McEldoon, J.P., Clark, D.S. and J.S. Dordick. 1999. Oxidation of polycyclic aromatic hydrocarbons catalyzed by soybean peroxidise. *Applied Biochemical Biotechnology* 80 (3): 221–230.

Lampela, K. and K. Jolma. 2011. Mechanical oil spill recovery in ice: Finnish approach. In: *Proceedings of the 2011 International Oil Spill Conference Proceedings (IOSC).* American Petroleum Institute. p. abs189.

Leahy, J.G. and R.R. Colwell. 1990. Microbial Degradation of hydrocarbons in the environment. *Microbial Reviews* 53(3): 305-315.

Lee, K. 2000. In situ bioremediation of oiled shoreline environments. Opportunities for Advancement of Environmental Applications of Marine Biotechnology. In: *Proceedings of the October 5-6, 1999, Workshop. The National Research Council of the National Academy of Sciences and the National Academy of Engineering.* National Academy Press, Washington, DC. pp. 44-60. http://www.nap.edu/read/9988/chapter/9.

Lee, K. and E.M. Levy. 1991. Bioremediation: Waxy crude oils stranded on low-energy shorelines. In: *Proceedings of the 1991 Oil Spill Conference, American Petroleum Institute, Washington, D. C.* pp. 541-547.

Lee, K., Lunel, T., Wood, P., Swannell, R. and P. Stoffyn-Egli. 1997a. Shoreline clean up by acceleration of clay-oil flocculation processes. In: *Proceedings of the 1997 International Oil Spill Conference (Prevention, Behaviour, Control and Cleanup*. Fort Lauderdale, Florida, April 7-10, 1997. pp. 235-240.

Lee, K., Stoffyn-Egli, P., Tremblay, G.H., Owens, E.H., Sergy, G.A., Guénette, C.C. and R.C. Prince. 2003b. Oil-mineral aggregate formation on oiled beaches: Natural attenuation and sediment relocation. *Spill Science and Technology Bulletin* 8: 285-296.

Lee, K., Nedwed, T., Prince, R.C. and D. Palandro, 2013. Lab tests on the biodegradation of chemically dispersed oil should consider the rapid dilution that occurs at sea. *Marine Pollution Bulletin* 73: 314–318.

Li, P., Chen, B., Li, Z.L., Zheng, X., Wu, H.J., Jing, L. and K. Lee. 2014a. A Monte Carlo simulation based two-stage adaptive resonance theory mapping approach for offshore oil spill vulnerability index classification. *Marine Pollution Bulletin* 86(1-2): 434-442.

Li, P., Chen, B., Zhang, B.Y., Jing, L. and J.S. Zheng. 2014b. Monte Carlo simulation-based dynamic mixed integer nonlinear programming for supporting oil recovery and devices allocation during offshore oil spill responses. *Ocean & Coastal Management* 89: 58-70.

Li, Z., Lee, K., King, T., Boufadel, M.C. and A.D. Venosa. 2008. Assessment of chemical dispersant effectiveness in a wave tank under regular nonbreaking and breaking wave conditions. *Marine Pollution Bulletin* 56(5): 903–912.

Li, Z., Lee, K., King, T., Boufadel, M.C. and A.D. Venosa. 2009. Evaluating chemical dispersant efficacy in an experimental wave tank: 2, significant factors determining in-situ oil droplet size distribution. *Environmental Engineering Science* 26(9): 1407-1418.

Lunel, T., Lee, K., Swannell, R.,Wood, P., Rusin, J., Bailey, N., Halliwell, C., Davies, L., Sommerville, M., Dobie, A., Mitchell, D. and M. McDonagh. 1996. Shoreline clean up during the Sea Empress Incident: The role of surf washing (clay-oil flocculation), dispersants and bioremediation. In: *Proceedings of the 19th Arctic and Marine Oilspill Program (AMOP) Technical Seminar, June 12-14, 1996, Calgary, Alberta, Canada.* Environment Canada. pp. 1521-1540.

Macek, T., Mackova, M. and J. Kas. 2000. Exploitation of plants for the removal of organics in environmental remediation. *Biotechnology Advances* 18: 23–34.

Macnaughton, S.J., Stephen, J.R., Venosa, A.D., Davis, G.A. Chang, Y-J. and D.C. White. 1999. Microbial population changes during bioremediation of an experimental oil spill. Applied Environmental Microbiology 65: 3566-3574.

Maki, H., Sasaki, T. and S. Harayama. 2001. Photo-oxidation of biodegraded crude oil and toxicity of the photo-oxidized products. *Chemosphere* 44(5): 1145-1151.

McGenity, T.J., Folwell, B.D., McKew, B.A. and G.O. Sanni. 2012. Marine crude-oil biodegradation: A central role for interspecies interactions. *Aquatic Biosystems* 8:10.

McNutt, M.K., Camilli, R., Crone, T.J., Guthrie, G.D., Hsieh, P.A., Ryerson, T.B., Savas, O. and F. Shaffer. 2012. Review of flow rate estimates of the Deepwater Horizon oil spill. *Proceedings of the National Academy of Sciences of the United States of America* 109 (50): 20260-20267.

Merlin, F.X. and P. Le Guerroué. 2009. Use of Sorbents for Spill Response: Operational Guide. Cedre. http://www.commissionoceanindien.org/fileadmin/resources/Autoroute%20maritime%20CEDRE/Cedre%20Use%20of%20Sorbents%20for%20Spill%20Response%20Cedre.pdf

ED_001324_00007647-00433

Michaels, D. and J. Howard. 2012. Review of the OSHA-NIOSH response to the Deepwater Horizon oil spill: Protecting the health and safety of cleanup workers. *PLoS Currents* 2012 Jul 18. Edition 1. doi: 10.1371/4fa83b7576b6e.

Michel, J. and B. Benggio. 1999. Guidelines for selecting appropriate cleanup endpoints at oil spills. In: *Proceedings of the 1999 International Oil Spill Conference*. American Petroleum Institute. pp. 591-595.

Michel, J., Keane, P. and B. Benggio. 2008. Pre-authorization for the use of solidifiers: results and lessons learned. In: Proceedings of the 2008 International Oil Spill Conference. American Petroleum Institute, Washington, D.C. pp. 345-348.

Mills, M.A., Bonner, J.S., Simon, M.A., McDonald, T.J. and R.L. Autenrieth. 1997. Bioremediation of a controlled oil release in a wetland. In: *Proceedings of the 20st Arctic and Marine Oilspill Program (AMOP) Technical Seminar*. Environment Canada, Ottawa. Pp. 606-616.

Mortazavi, B., Horel, A., Beazley, M. J. and P.A. Sobecky. 2013. Intrinsic rates of petroleum hydrocarbon biodegradation in Gulf of Mexico intertidal sandy sediments and its enhancement by organic substrates. *Journal of Hazardous Materials* 244: 537-544.

Muratova, A.Y., Kapitonova, V.V., Chernyshova, M.P. and O.V. Turkovskaya. 2009. Enzymatic Activity of Alfalfa in a Phenanthrene-contaminated Environment. *World Academy of Science, Engineering and Technology* 58: 569-574.

Muschenheim, D.K. and K. Lee, 2003. Removal of oil from the sea surface through particulate interactions: Review and prospectus. *Spill Science and Technology Bulletin* 8: 9-18.

Myers, D. 2006. Surfactant Science and Technology. Third Edition. John Wiley and Sons, Inc., Hoboken, NJ.

NOAA (National Oceanic & Atmospheric Administration). 1992. Shoreline Countermeasure Manual. Seattle, Washington.

NOAA. 2007. In-Situ Burning. Office of Response and Restoration. http://response.restoration.noaa.gov/oil-and-chemical-spills/oil-spills/resources/in-situ-burning.html.

NOAA. 2015. Spill Containment Methods. Office of Response and Restoration. http://response.restoration.noaa.gov/oil-and-chemical-spills/oil-spills/spill-containment-methods.html

NOAA and API (American Petroleum Institute). 1994. Options for Minimizing Environmental Impacts of Freshwater Spill Response.

NRC (National Research Council). 1989. Using Oil Spill Dispersants on the Sea. National Academy Press, Washington, DC.

NRC. 2014. Responding to Oil Spills in the U.S. Arctic Marine Environment. National Academies Press, Washington, D.C. http://www.nap.edu/catalog/18625/responding-to-oil-spills-in-the-us-arctic-marine-environment. (Accessed June, 2015)

Nuka Research and Planning Group. 2010. Oil Spill Prevention and Response in the U.S. Arctic: Unexamined Risks, Unacceptable Consequences. U.S. Arctic Program, Pew Environment Group.

Oskins, C.J. and D. Bradley. 2005. "Extreme" cold weather oil spill response techniques & strategies - ice & snow environments. In: *Proceedings of the 2005 International Oil Spill Conference.* American Petroleum Institute. pp. 521-525.

Pelletier, É. and R. Siron. 1999. Silicone-based polymers as oil spill treatment agents. *Environmental Toxicology and Chemistry* 18: 813-818.

Pezeshki, S.R., Hester, M.W., Lin, Q. and J.A. Nyman. 2000. The effects of oil spill and clean-up on dominant US Gulf coast macrophytes: a review. *Environmental Pollution* 108: 129-139.

Pilipenko, N., Lerondeau, B. and E. Vial. 2013. Oil spill response in the arctic. In: Proceedings of the 2013 *SPE Arctic and Extreme Environments Technical Conference and Exhibition.* Society of Petroleum Engineers.

Porter, M.R. 1991. Handbook of Surfactants. Blackie and Sons, Ltd., Glasgow, Scotland.

Powter, C.B. (Compiler). 2002. Glossary of Reclamation and Remediation Terms Used in Alberta – 7th Edition. Alberta Environment, Science and Standards Branch, Edmonton, AB. Pub. No. T/655; Report No. SSB/LM/02-1. http://environment.gov.ab.ca/info/library/6843.pdf

ED_001324_00007647-00434

Prince, R.C., Gramain, A. and T.J. McGenity. 2010. Prokaryotic hydrocarbon degraders. In: Timmis, K.N., McGenity, T.J., Meer, J.R. and V. Lorenzo (eds.) *Handbook of Hydrocarbon and Lipid Microbiology.* Springer, Berlin Heidelberg. pp. 1671-1692.

Ramsay, M.A., Swannell, R.P.J., Shipton, W.A., Duke, N.C. and R.T. Hill. 2000. Effect of bioremediation on the microbial community in oiled mangrove sediments. *Marine Pollution Bulletin* 41: 413-419.

Region 10 Regional Response Team and the Northwest Area Committee. 2015. 9301. Oil spill best management practices. In: *NW Area Contingency Plan*. RRT/NWAC, Seattle, WA. http://www.rrt10nwac.com/nwacp/

Rosenberg, E. and E.Z. Ron. 1996. Bioremediation of petroleum contamination. In: Crawford, R.L. and D.L. Crawford (eds.) *Bioremediation: Principles and Applications*. Cambridge University Press, UK. Pp. 100-124.

Salt, D., Hughes, E. and S. Gair. 2012. Monitoring and quantifying oil pollution using aerial surveillance~ problems and solutions. In: *Proceedings of the 2012 SPE International Conference on Health, Safety and Environment in Oil and Gas Exploration and Production*. Society of Petroleum Engineers.

Schmidt, B., Meyer, P. and S. Potter. 2014. Testing of oil recovery skimmers in ice at Ohmsett The National Oil Spill Response Research & Renewable Energy Test Facility. In *Proceedings of the 2014 International Oil Spill Conference Proceedings*. American Petroleum Institute. pp. 618-633.

Schulze, R. and H.L. Hoffman. 1993. World Catalog of Oil Spill Response Products. Fourth Edition. Elkride, MD.

Sergy, G. A., and E.H. Owens. 2007. Guidelines for Selecting Shoreline Treatment Endpoints for Oil Spill Response. Emergency Science and Technology Division, Environment Canada, Ottawa, ON.

Sergy, G. A. and E.H. Owens. 2008. Selection and use of shoreline treatment endpoints for oil spill response. In: *Proceedings of the 2008 International Oil Spill Conference Proceedings*. American Petroleum Institute. pp. 435-441.

Serova, I. 1992. Behaviour of oil in ice and water at low temperatures. In: *Combating Marine Oil Spills in Ice and Cold Climates: HELLCOM Seminar, Helsinki, Finland.*

Shin, W.S., Tate, P.T., Jackson, W.A. and J.H. Pardue. 1999. Bioremediation of an experimental oil spill in a salt marsh. In: Means, J.L. and R.E. Hinchee (eds.) *Wetlands and Remediation: An International Conference*. Battelle Press, Columbus, OH. pp. 33-40.

Shin, W.S., Pardue, J.H. and W.A. Jackson. 2000. Oxygen demand and sulfate reduction in petroleum hydrocarbon contaminated salt marsh soils. *Water Research* 34(4): 1345-1353.

Sundaravadivelu, D., Suidan, M.T. and A.D. Venosa. 2015. Parametric study to determine the effect of temperature on oil solidifier performance and the development of a new empirical correlation for predicting effectiveness. *Marine Pollution Bulletin* 95(1): 297-304.

Transport Canada. 2013. A Review of Canada's Ship-Source Oil Spill Preparedness and Response Regime: Setting the Course for the Future. Tanker Safety Panel Secretariat, Ottawa. www.tc.gc.ca/eng/tankersafetyexpertpanel/menu.htm. (accessed June 16 2015).

Tyagi, M., da Fonseca, M. M. R. and C.C. de Carvalho. 2011. Bioaugmentation and biostimulation strategies to improve the effectiveness of bioremediation processes. *Biodegradation* 22(2): 231-241.

U.S. EPA. 1999. Understanding Oil Spills and Oil Spill Response. EPA 540-K-99-007. Office of Emergency and Remedial Response, U.S. Environmental Protection Agency.

US Coast Guard. 2001. Characteristics of Response Strategies: A Guide for Spill Response Planning in Marine Environments. A joint publication of the American Petroleum Institute, National Oceanic and Atmospheric Administration, U.S. Coast Guard, and U.S. Environmental Protection Agency.

Valentine, D.L., Fisher, G.B., Bagby, S.C., Nelson, R.K., Reddy, C.M., Sylva, S.P. and M.A. Woo. 2014. Fallout plume of submerged oil from *Deepwater Horizon. Proceedings of the National Academy of Sciences (USA)* 111(45): 15906-15911.

Venosa, A.D. and X. Zhu. 2005. Guidance for the bioremediation of oil-contaminated wetlands, marshes, and marine shorelines. In: Fingerman, S. and R. Nagabhushanam (eds.) *Bioremediation of Aquatic and Terrestrial Ecosystems*. Science Publishers, Inc. Enfield, NH. pp. 141-172.

Venosa, A.D., Suidan, M.T., Wrenn, B.A., Strohmeier, K.L., Haines, J.R., Eberhart, B.L., King, D. and E. Holder. 1996. Bioremediation of an experimental oil spill on the shoreline of Delaware Bay. *Environmental Science and Technology* 30(5): 1764-1775.

Venosa, A.D., Lee, K., Suidan, M.T., Garcia-Blanco, S., Cobanli, S., Moteleb, M., Haines, J.R., Tremblay, G. and M. Hazelwood. 2002. Bioremediation and biorestoration of crude oil-contaminated freshwater wetland on the St. Lawrence River. *Bioremediation Journal* 6(3): 261-281.

Venosa, A.D., Anastas, P.T., Barron, M.G., Conmy, R.N., Greenberg, M.S. and G.J. Wilson. 2012. Science-based decision making on the use of dispersants in the Deepwater Horizon oil spill: EPA's perspectives. In: Somasundaran, P., Farinato, R., Patra, P. and K. Papadopoulos (eds.) *Oil Spill Remediation: Colloid Chemistry-Based Principles and Solutions.* John Wiley and Sons, Inc.

Verma, M., Gendreau, M. and G. Laporte. 2013. Optimal location and capability of oil-spill response facilities for the south coast of Newfoundland. *Omega* 41(5): 856-867.

Wadsworth, T. 1995. Containment & Recovery of Oil Spills at Sea. Methods and Limitations. International Tanker Owners Pollution Federation Limited. http://www.itopf.com/knowledge-resources/documents-guides/document/containment-recovery-of-oil-spills-at-sea-methods-and-limitations/).

Walker, A.H., Kucklick, J.H. and J. Michel. 1999. Effectiveness and environmental considerations for non-dispersant countermeasures. *Pure Applied Chemistry* 71: 67-81.

Wang, Z. and M. Fingas. 2003. Development of oil hydrocarbon fingerprinting and identification techniques. *Marine Pollution Bulletin* 47: 423–452.

Wang, Y. Z., Yan, Q. B., Huang, J. J., Li, B., and J.L. Gao. 2012. Recovery of lubricating oil for cold strip rolling. *Applied Mechanics and Materials* 189: 44-47.

Webler, T. and F. Lord. 2010. Planning for the human dimensions of oil spills and spill response. *Environmental Management* 45(4): 723–738.

Widada, J., Nojiri, H., Kasuga, K., Yoshida, T.,, Habe, H. and T. Omori. 2002. Molecular detection and diversity of polycyclic aromatic hydrocarbon-degrading bacteria isolated from geographically diverse sites. *Applied Microbiology and Biotechnology* 58: 202–209.

Wilkman, G., Ritari, E., Uuskallio, A. and M. Niini. 2014. Technological Development in Oil Recovery in Ice Conditions. In: *OTC Arctic Technology Conference. Offshore Technology Conference.*

Wrenn, B.A., Virkus, A., Mukherjee, B. and A.D. Venosa. 2009. Dispersibility of crude oil in freshwater. *Environmental Pollution* 157: 1807-1814.

Yasin, G., Bhanger, M. I., Ansari, T. M., Naqvi, S. M. S. R., Ashraf, M., Ahmad, K. and F.N. Talpur. 2013. Quality and chemistry of crude oils. *Journal of Petroleum Technology and Alternative Fuels* 4(3): 53-63.

Zhong, Z., and F. You. 2011. Oil spill response planning with consideration of physicochemical evolution of the oil slick: A multiobjective optimization approach. *Computers & Chemical Engineering* 35(8): 1614-1630.

Zhu, X., Venosa, A.D., Suidan M.T. and K. Lee. 2001. Guidelines for the Bioremediation of Marine Shorelines and Freshwater Wetlands. U.S. Environmental Protection Agency

Zhu, X., Venosa, A.D., Suidan, M.T. and K. Lee. 2004. Guidelines for the Bioremediation of Oil-Contaminated Salt Marshes. EPA/600/R-04-074. U.S. Environmental Protection Agency.

Chapter 7

Akinyokun, O. C. and U.G. Inyang. 2013. Experimental study of neuro fuzzy -genetic framework for oil spillage risk management. *Artificial Intelligence Research* 2(4): 13-35.

ACAP. 2008. The Atlantic Coastal Action Program (ACAP) – Year in Review 2006/07. Atlantic Coastal Action Program, Environment Canada, Dartmouth, NS.

ADEC. 1993. The Exxon Valdez Oil Spill -Final Report. Department of Environ mental Conservation (ADEC) Final Report. State of Alaska Response.

ADEC. 2015. Oil and Other Hazardous Substances Pollution Control. 18 AAC 75, Department of Environmental Conservation. Alaska.

Adger, W. N. 2006. Vulnerability. *Global Environmental Change* 16(3): 268-281.

ED_001324_00007647-00436

Adler, E., and M. Inbar. 2007. Shoreline sensitivity to oil spills, the Mediterranean coast of Israel: assessment and analysis. *Ocean & Coastal Management* 50(1): 24-34.

AER (Alberta Energy Regulator). 2015. Compliance Dashboard. http://www1.aer.ca/compliancedashboard/incidents.html.

Aguilera, F., Méndez, J., Pásaro, E. and B. Laffon. 2010. Review on the effects of exposure to spilled oils on human health. *Journal of Applied Toxicology* 30(4): 291-301.

Allan, S. E., Smith, B. W. and K.A. Anderson. 2012. Impact of the Deepwater Horizon oil spill on bioavailable polycyclic aromatic hydrocarbons in Gulf of Mexico coastal waters. *Environmental Science & Technology* 46(4): 2033-2039.

Allen, B., Stokey, R., Forrester, N., Gouldsborough, R., Purcell, M. and C. von Alt. 1997. Remus: a small, low cost AUV; system description, field trials and performance results. In: *Proceedings of IEEE/MTS Oceans Conference, October 1997, volume 2*. Pp. 994–1000.

Anderson, E. J. and D.J. Schwab. 2012. Contaminant transport and spill reference tables for the St. Clair River. *Marine Technology Society Journal* 46(5): 34-47.

Andrade, F., Lutz, G., Lyra, A. P., Ranieri, A., and B. Lemos. 2013. Oil Spill R esponse Equipment and Techniques for Waxy Oils in Brazil. In: *Proceedings of the SPE Latin -American and Caribbean Heath, Safety, Environment and Social Responsibility Conference* . Society of Petroleum Engineers.

AOSRT JIP, 2014. Environmental Impacts of Ar ctic Oil Spills and Arctic Spill Response Technologies - Literature Review and Recommendations. December 2014. Arctic Oil Spill Response Technology Joint Industry Programme.

API (American Petroleum Institute). 2009. Analysis of US Spillage. API Publicatio n No. 356. American Petroleum Institute, Washington, D.C.

Baars, B. J. 2002. The wreckage of the oil tanker 'Erika' —human health risk assessment of beach cleaning, sunbathing and swimming. *Toxicology Letters* 128(1): 55-68.

Barenboim, G. and A.J. Saveka. 2 012. New methodological aspects of assessment of biological hazard of oil spills. In: *Proceedings of the 35th AMOP Technical Seminar on Environmental Contamination and Response, Vancouver, Canada*. Environment Canada, Ottawa. Pp. 13-30.

Barenboim, G., Saveka, A.J. and O. Avandeeva. 2013. Development of a system for the early detection and monitoring of oil spills on water bodies with a glance to its use in the Arctic Zone. In: *Proceedings of the 36th AMOP Technical Seminar on Environmental Contamination and Response, Halifax, Canada*. Environment Canada, Ottawa.

Barron, M. G. 2012. Ecological impacts of the deepwater horizon oil spill: Implications for immunotoxicity. *Toxicologic Pathology* 40(2): 315-320.

Bejarano, A. C. and A.J. Mearns. 2015. Improving envir onmental assessments by integrating Species Sensitivity Distributions into environmental modeling: Examples with two hypothetical oil spills. *Marine Pollution Bulletin* 93(1): 172-182.

Benoit, L.E. 2007. Rail Transport and the Environment in Canada: A Study Prepared in Support of the 2007 Railway Safety Act Review. Benoit & Associates. Report prepared for Railway Safety Act Review Advisory Panel.

Board of Ocean Studies. 2014. Reducing Coastal Risk on the East and Gulf Coasts. National Academies Press, Washington, DC.

Board of Ocean Studies and Marine Board. 2014. Responding to Oil Spills in the US Arctic Marine Environment. The National Academies Press, Washington, DC.

Boehm, P. D., Page, D. S., Brown, J. S., Neff, J. M., and E. Gundlach. 2014. Long -term fate and persistence of oil from the Exxon Valdez Oil Spill: Lessons learned or history repeated? In: *Proceedings of the 2014 International Oil Spill Conference*. American Petroleum Institute. pp. 63-79.

Bostrom, A., Walker, A. H., Scott, T., Pavia, R., Leschin e, T. M., and K. Starbird. 2015. Oil spill response risk judgments, decisions, and mental models: findings from surveying US stakeholders and coastal residents. *Human and Ecological Risk Assessment: An International Journal* 21(3): 581-604.

British Columbia Ministry of Environment. 2014. Land Based Spill Preparedness and Response in British Columbia. Ministry of Environment - Policy Intentions Paper for Consultation. British Columbia.

British Columbia Ministry of Environment. 2015. Queen of the North BC Ferr y Sinking. http://www.env.gov.bc.ca/eemp/incidents/2006/queen_north_06.htm. (accessed August 29, 2015)

Brown, C. and M. Fingas. 2003. Review of the development of laser fluoros ensors for oil spill application. *Marine Pollution Bulletin* 47: 477–484.

ED_001324_00007647-00437

Buist, I., Potter, S., and T. Nedwed. 2011. Herding agents to thicken oil spills in drift ice for in situ burning: new developments. In: *Proceedings of the 2011 International Oil Spil l Conference, May 23 -26, 2011, Portland, Oregon, USA*. American Petroleum Institute. Pp. abs230.

Bush, B. J. 2011. Addressing the regulatory collapse behind the deepwater horizon oil spill: Implementing a best available technology regulatory regime for deep water oil exploration safety and cleanup technology. *Journal of Environmental Law and Litigation* 26: 535-570.

Cai, Q.H., Zhang, B.Y., Chen, B., and T. Cao. 2014b. Biodispersant produced by a Rhodococcus

Cai, W.X., Zheng, Y.W., Shi, Y.Q., and H.L. Zhong. 2 009. Threat Level Forecast for Ship's Oil Spill - Based on BP Neural Network Model. In: *Proceedings of the International Conference on Computational Intelligence and Software Engineering, 2009 (*CiSE 2009). IEEE. pp. 1-4.

Canadian National Railway Company (CN). 2015. Gogama, Ontario: Get the Facts. http://www.cngogamainfo.ca/en/index.asp.

Cao, X and S. Fan. 2013. The synthetic assessment modeling of ships' oil spill risk based on fuzzy neural network. In: *Proceedings of International Conference on Intelligent Systems and Applications (ISA). Wuhan, China. 2013 May 22-23*. Pp. 368-371.

CAPP. 2015. Infrastructure and Transportation: Pipelines. http://www.capp.ca/canadian-oil-and-natural-gas/infrastructure-and-transportation/pipelines. (accessed August 24, 2015).

CBC. 2005. Province unprepared for Wabamun spill: Klein. *CBC News* , Aug 11, 2005. http://www.cbc.ca/news/canada/province-unprepared-for-wabamun-spill-klein-1.542144

Chang, S. E., Stone, J., Demes, K., and M. Piscitelli. 2014. Consequences of oil spills: A review and framework for informing planning. *Ecology and Society* 19(2): 26.

Chapin III, F. S., Peterson, G., Berkes, F., Callaghan, T. V., Angelstam, P., Apps, M. an d C. Beier. 2004. Resilience and vulnerability of northern regions to social and environmental change. *AMBIO: A Journal of the Human Environment* 33(6): 344-349.

Chapman, H., Purnell, K. and R.J. Law. 2007. The use of chemical dispersants to combat oil spills at sea: A review of practice and research needs in Europe. *Marine Pollution Bulletin* 54(7): 827-838.

Chatterjee, S. 2015. Oil spill cleanup: Role of environmental biotechnology. In: *Applied Environmental Biotechnology: Present Scenario and Future Trends*. The Springer Press.

Chen, B., Zhang, B.Y., Li, P., Cai, Q.H., Lin, W.Y., and B. Liu. 2012. From Challenges to Opportunities: Towards Future Strategies and a Decision Support Framework for Oil Spill Preparedness and Response in Offshore Newfoundland and L abrador. Technical Report for the Harris Centre, St. John's, Canada.

Christoph, M., Vis, M. A., Rackliff, L. and H. Stipdonk. 2013. A road safety performance indicator for vehicle fleet compatibility. *Accident Analysis & Prevention* 60: 396-401.

Christopherson, S. and K. Dave. 2014. A New Era of Crude Oil Transport: Risks and Impacts in the Great Lakes Basin. Community and Regional Development Institute, Department of Development Sociology, Cornell University. https://cardi.cals.cornell.edu/sites/cardi.cals.cornell.edu/files/shared/documents/CardiReports/A-New-Era-of-Crude-Oil-Transport.pdf.

Chung, P. P., Balla rd, J. W. O. and R.V. Hyne. 2012. It's All in the Genes: How Genotype Can Impact Upon Response to Contaminant Exposure and the Implications for Biomonitoring in Aquatic Systems. INTECH Open Access Publisher.

Cleveland, C. 2010. The Challenges of Oil Spill Response in the Arctic. National Commission on the BP Deepwater Horizon Oil Spill and Offshore Drilling, US.

Cluzel, F., Yannou, B., Leroy, Y. and D. Millet. 2012. Proposition of an adapted management process to evolve from an unsupervised LCA of complex industrial systems towards an eco -designing organisation. *Concurrent Engineering: Research and Applications* 20(2): 111-126.

Coelho, G., Clark, J. and D. Aurand. 2013. Toxicity testing of dispersed oil requires adherence to standardized protocols to assess potential real world effects. *Environmental Pollution* 177: 185-188.

Coolbaugh, T., DeMicco, E. and E. Kennedy. 2014. Dispersant -related oil spill response communication tools: Toward an enhanced approach to conveying complex topics in an approachable manner. In: *Proceedings of the 2014 International Oil Spill Conference* . A merican Petroleum Institute. pp. 1163-1171.

Cox, R. 2014. Risk assessment and planning for offshore oil spill response preparedness. In: *SPE International Conference on Health, Safety, and Environment*. Society of Petroleum Engineers.

ED_001324_00007647-00438

Crosby, S., Fay, R., G roark, C., Kani, A., Smith, J.R., Sullivan, T., and R. Pavia. 2013. Transporting Alberta Oil Sands Products: Defining the Issues and Assessing the Risks. NOAA Technical Memorandum NOS OR&R 44. NOAA Office of Response and Restoration, Seattle.

Dalbey, B. 20 15. Oil Company Agrees to Mackinac Straits Heavy Crude Oil Ban. September 3, 2015, Patch Network. http://patch.com/michigan/wyandotte/oil-company-agrees-mackinac-straits-heavy-crude-oil-ban-0. [accessed September 18, 2015]

Daling, P. S., Leirvik, F., Almås, I. K., Brandvik, P. J., Hansen, B. H., Lewis, A. and M. Reed. 2014. Surface weathering and dispersibility of MC252 crude oil. *Marine Pollution B ulletin* 87(1): 300-310.

Davies, A. J., and M.J. Hope. 2015. Bayesian inference -based environmental decision support systems for oil spill response strategy selection. Marine Pollution Bulletin: doi:10.1016/j.marpolbul.2015.05.041.

DeCola, E., Robertson, T., Fletcher, S., and S. Harvey. 2006. Offshore Oil Spill Response in Dynamic Ice Conditions. A Report to WWF on Considerations for the Sakhalin II Project, Alaska. Nuka Research.

DeMicco, E., Schuler, P. A., Omer, T. and B. Baca. 2011. Net environmental ben efit analysis (NEBA) of dispersed oil on nearshore tropical ecosystems: Tropics –the 25th year research visit. In: *Proceedings of the 2011 International Oil Spill Conference* . American Petroleum Institute. p. abs282.

Department of Attorney General. 2015. Mic higan Petroleum Pipeline Task Force Report. Department of Attorney General and Department of Environmental Quality, State of Michigan. July 2015. Lansing, USA. http://www.michigan.gov/documents/deq/M_Petroleum_Pipeline_Report_2015-10_reducedsize_494297_7.pdf

Dix, M., Kolodziejski, A. and J. Owens. 2011. 19 down, 34 nations to go: AFRICOM and spill response capacity building. In: *Proceedings of the 2011 International Oil Spill Conference* . American Petroleum Institute. p. abs134.

Doerffer, J. W. 2013. Oil Spill Response in the Marine Environment. Elsevier.

Dollhopf, R. and M. Durno. 2011. Kalamazoo River \Enbridge Pipeline Spill 2010. In: *Proceedings of the 2011 International Oil Spill Conference*. American Petroleum Institute. p. abs422.

Dollhopf, R. H., Fitzpatrick, F. A., Kimble, J. W., Capone, D. M., Graan, T. P., Zelt, R. B., and R. Johnson. 2014. Response to heavy, non -floating oil spilled in a Great Lakes river environment: a multiple -lines-of-evidence approach for submerged oil assessment and recovery. In: *Proceedings of the 2014 International Oil Spill Conference Proceedings*. American Petroleum Institute. pp. 434-448.

Echavarria-Gregory, M. A ., and J.D. Englehardt. 2015. A predictive Bayesian data -derived multi -modal Gaussian model of sunken oil mass. *Environmental Modelling & Software* 69: 1-13.

Ecosocialist Network. 2013. Lac-Mégantic: A Social and Ecological Tragedy. The Bullet No. 856.

El-Zahaby A.M., Kabeel A.E., Bakry A.I., and A.M. Khaira. 2011. Effect of disk offset angle on the hydrodynamic performance of rotating disk skimmers during oil spill recovery. *Environmental Engineering Science* 28(2): 113-119.

Environment Canada. 1992. Oilspill SCAT Manual for the Coastlines of British Columbia. Environment Canada, Technology Development Branch, Edmonton, AB, Regional Programme Report 92 -03, 245 pp.

Environment Canada. 1999. National Environmental Emergencies Contingency Plan. Environmen tal Emergencies Program. Environment Canada. No. de cat. En40-584/1999E.

Environment Canada. 2015a. Canadian Aquatic Biomonitoring Network (CABIN). http://www.ec.gc.ca/rcba-cabin. (accessed June 21, 2015).

Environment Canada. 2015b. Environmental Effects Monitoring. http://ec.gc.ca/esee-eem/default.asp?lang=En&n=4B14FBC1-1. (accessed June 21, 2015).

Environment Canada. 2015c. Proceedings of t he 38th Arctic and Marine Oilspill Program (AMOP) Technical Seminar on Environmental Contamination and Response, June 2-4, Vancouver, Canada. Environment Canada, Ottawa.

ERCB (Energy Resources Conservation Board). 2013. Plains Midstream Canada ULC NPS 20 Pipeline Failure Licence No. 5592, Line No. 1 April 28, 2011. ERCB Investigation Report. https://www.aer.ca/documents/reports/IR_20130226-PlainsMidstream.pdf.

Ermakov, S.A. 2013. Oil slicks on the sea surface and their remote sensing. In: *Proceedings of the 36th AMOP Technical Seminar on Environmental Contamination and Response, Halifax, NS* . Environment Canada, Ottawa. Pg, 539-547.

Etkin, D.S. 2012a. Cook Inlet Maritime Risk Assessment: Spill Baseline & Accident Casualty Study. Spill Scenarios and Impacts. A report prepared for the Glosten Associates.

Etkin, D.S. 2012b. Development of a Quantitative Risk Model for Inland Facility and Pipeline Spills. In: *Proceedings of t he 35th AMOP Technical Seminar on Environmental Contamination and Response*, Vancouver, Canada.

Etkin, D.S., Culpepper, D.B. and L. Martinez. 2013. Application of some remote sensing techniques for oil spill monitoring, response, and planning. In: *Proceedings of the 36th AMOP Technical Seminar on Environmental Contamination and Response, Halifax, Canada* . Environment Canada, Ottawa. Pp. 522-538.

Etkin, D.S., Reilly, T. and D. French -McCay. 2011. State -of-the-art risk -based approach to spill contingency planni ng and risk management. In: *Proceedings of the 34th AMOP Technical Seminar on Environmental Contamination and Response, Banff, Canada* . Environment Canada, Ottawa. Pp. 906-927.

ExxonMobil Corporation. 2013. Offshore Arctic Oil Spill Prevention, Preparedness and Response. http://www.exxonmobil.com/Corporate/Files/news_pub_2013-arctic-spill-prevent.pdf. (accessed June 15 2015).

Fernandes, R., Neves, R. and C. Viegas. 2013. Integration of an oil and inert spill model in a framework for risk management of spills at sea: A case study for the Atlantic area. In: *Proceedings of the 36th AMOP T echnical Seminar on Environmental Contamination and Response, Halifax, Canada.* Environment Canada, Ottawa. Pp. 326-353.

Ferraro, G., Meyer-Roux, S., Muellenhoff, O., Pavlih, M., Svetak, J., Tarchi, D. and K. Topouzelis. 2009. Long term monitoring of oil sp ills in European seas. *International Journal of Remote Sensing* 30(3): 627-645.

Fingas, M.F. (ed.). 2011. Oil Spill Science and Technology: Prevention, Response, and Cleanup. Elsevier/Gulf Professional Publication.

Fingas, M.F. 2012. Buoys and Devices for Oil Spill Tracking. In: *Proceedings of the 35th AMOP Technical Seminar on Environmental Contamination and Response*, Vancouver, Canada

Fitzpatrick, F., Bejarano, A., Michel, J. and L. Williams. 2012. Net Environmental Benefit Analysis (NEBA) Relative Risk Ra nking Conceptual Design Kalamazoo River System Enbridge Line 6B MP 608 Marshall, MI Pipeline Release. USGS, Research Planning, Inc., and US Fish and Wildlife Service. http://www.epa.gov/enbridgespill/ar/enbridge-AR-0963.pdf.

French-McCay, D. P. 2002. Development and application of an oil toxicity and exposure model, OilToxEx. *Environmental Toxicology and Chemistry* 21(10): 2080-2094.

Fritt-Rasmussen, J., Wegeberg, S., Boertmann, D., Clausen, D.S., Johansen, K.L., and A. Mosbech. 2013. Net Environmental Benefit Analysis as a Decision Tool for Greenlandic Oil Spill Response. In: *Proceedings of the 36th AMOP Technical Seminar on Environme ntal Contamination and Response, Halifax, Canada.* Environment Canada, Ottawa. Pp. 765-776.

Gala, W., Lipton, J., Cernera, P., Ginn, T., Haddad, R., Henning, M., Jahn, K., Landis, W., Mancini, E., Nicoll, J., Peters, V., and J. Peterson. 2009. Ecological ri sk assessment and natural resource damage assessment: Synthesis of assessment procedures. *Integrated Environmental Assessment and Management* 5(4): 515-522.

Gill, D.A., Ritchie, L.A., Picou, J.S., Langhinrichsen -Rohling, J., Long, M.A. and J.W. Shenesey. 20 14. The Exxon and BP oil spills: a comparison of psychosocial impacts. *Natural Hazards* 74(3): 1911-1932.

Gill, S.D., Bonke, C.A., and J. Carter. 1985. Management of the UNIACKE G-72 incident. In: *Proceedings of the 1985 International Oil Spill Conference*. American Petroleum Institute. pp. 311-314.

Goodman, R. 1994. Overview and future trends in oil spill remote sensing. *Spill Science & Technology Bulletin* 1(1): 11–21.

Government of Canada. 2010. Northern Canada Vessel Traffic Services Zone Regulations (SOR/ 2010-127). Annex of *Canada Shipping Act 2001*, Ottawa.

Graham, B., Reilly, W. K., Beinecke, F., Boesch, D. F., Garcia, T. D., Murray, C. A. and F. Ulmer. 2011. Deep Water: The Gulf Oil Disaster and the Future of Offshore Drilling. Report to the President, National Commission on the BP Deepwater Horizon Oil Spill and Offshore Drilling, Washington, DC.

Grattan, L.M., Roberts, S., Mahan Jr. W.T., McLaughlin, P.K., Otwell, W.S. and J.G. Morris Jr. 2011. The early psychological impacts of the Deepwater Horizon o il spill on Florida and Alabama communities. *Environ Health Perspectives* 119:838-843.

GRCA. 2015. Grand River Conservation Authority. http://www.grandriver.ca. (accessed June 21, 2015).

Griffiths, G., Stevenson, P. , Webb, A. T., Millard, N. W., McPhail, S. D., Pebody, M. and J. R. Perrett. 1999. Open ocean operational experience with the autosub -1 autonomous underwater vehicle. In: *Proceedings of the 11th International Symposium on Unmanned Untethered Submersible Technology, Durham, NH, USA.*

Hausberger, O., Hoegn, L. A. and K. Soliman. 2012. Management decision matrix for shale gas projects in Europe. In: *SPE Hydrocarbon Economics and Evaluation Symposium* . Society of Petroleum Engineers. Pp. 545-556

Hodges, B. R., Orfila, A., Sayol, J. M., and X. Hou. 2015. Operational oil spill modelling: from science to engineering applications in the presence of uncertainty. In: **Ehrhardt,** M. (ed.) *Mathematical Modelling and Numerical Simulation of Oil Pollution Problems* . Springer I nternational Publishing. pp. 99-126.

Hollnagel, E. and D.D. Woods. 2006. Epilogue: Resilience engineering precepts. In: Hollnagel, E., Woods, D.D. and N. Leveson (eds.) *Resilience Engineering–Concepts and Precepts.* Ashgate Publishing Company, Aldershot. UK. Pp. 347-358.

Holt, T. J., D. R Jones, S. J. Hawkins, and R. G. Hartnoll. 1995. The sensitivity of marine communities to man-induced change: a scoping report. Contract science report 65. Maes -y-Ffynnon, Penrhosgarnedd, Bangor, Wales.

Hopkins, A. 2000. Lessons from Longford. The Esso Gas Explosion. CCH Australia limited, Sydney.

Hou, X., and B.R. Hodges. 2014. Visualizing hydrodynamic uncertainty in operational oil spill modeling. In: *Proceedings of the 2014 International Oil Spill Conference* . American Petroleum Institute. p. 299013.

Hout, T., Parkin, T., Smith, S., and R. Ward. 2014. Combining best available technology: A systems approach for effective mechanical oil spill response. In: *Proceedings of the 2014 International Oil Spill Conference.* American Petroleum Institute. p. 300184

Howden, S. D., Barrick, D., and H. Aguilar. 2011. Applications of high frequency radar for emergency response in the coastal ocean: utilization of the Central Gulf of Mexico Ocean Observing System during the Deepwater Horizon oil spill and vessel tracking. In: Hou, W.W. and R. Armone (eds.) *Proceedings of SPIE 8030 Ocean Sensing and Monitoring III, 80300O.* International Society for Optics and Photonics.

Huang, G.H., Sae -Lim, N., Liu, L., and Z. Chen. 2001. An interval -parameter fuzzy -stochastic programming approach for municipal solid waste management and planning. *Environmental Modeling and Assessment* 6: 271-283.

Ihaksi, T., Kokkonen, T., Helle, I., Jolma, A., Lecklin, T., and S. Kuikka. 2011. Combining conservation value, vulnerability, and effectiveness of mitigation actions in spatial conservation decisions: An application to coastal oil spill combating. *Environmental Management* 47(5): 802-813.

IMO. 1990. International Convention on Oil Pollution Preparedness, Response and Co-operation, 1990, and Final Act of the Conference. International Maritime Organization. International Legal Materials. Vol. 30, No. 3 (MAY 1991), pp. 733 -761. Published by American Society of International Law. 29pp.

Inyang, U. G., and O. C. Akinyokun. 2014. A hybrid knowledge discov ery system for oil spillage risks pattern classification. *Artificial Intelligence Research* 3(4): 77.

IPIECA (International Petroleum Industry Environmental Conservation Association). 2000. Choosing Spill Response Options to Minimize Damage Net Environmental Benefit Analysis. London, UK.

IPIECA. 2015. Oil Spill Preparedness and Response: An Introduction. International Association of Oil & Gas Producers Report 520. International Petroleum Industry Environmental Conservation Association, London.

ITOPF. 2013. C ountry Profiles: Canada. A Summary of Oil Spill Response Arrangements. International Tanker Owners Pollution Federation. http://www.itopf.com/fileadmin/data/Documents/Country_Profiles/canada.pdf.

ITOPF. 2015. Oil Tanker Spill Statistics 2014. http://www.itopf.com/knowledge-resources/data-statistics/statistics/. (accessed August 24, 2015).

Jennings, K. and F. Schulberg. 2009. Guidance on developing safety performance indicators. *Process Safety Progress* 28(4): 362-366.

Jing, L., Chen, B., Zhang, B.Y. and P. Li. 2012. A stochastic simulation -based hybrid interval fuzzy programming approac h for optimizing the treatment of recovered oily water. *Journal of Ocean Technology* 7(4): 59-72.

Jing, L., Chen, B., Zhang, B.Y. and P. Li. 2015. Process simulation and dynamic control for marine oily wastewater treatment using UV irradiation. *Water Research* 81: 101-112.

ED_001324_00007647-00441

Johnson, E. I. 2014. Louisiana's Coastal Stewardship Program: Beach-nesting Bird Protection, Monitoring, and Community Outreach. Final Report. National Audubon Society, Baton Rouge, LA.

JRP (Joint Review Panel). 2013a. Connections: Report of the Joint Review Panel for the Enbridge Northern Gateway Project, Vol. 1. National Energy Board, Calgary, AB.. http://gatewaypanel.review-examen.gc.ca/clf-nsi/dcmnt/rcmndtnsrprt/rcmndtnsrprtvlm1-eng.pdf.

Judson, R. S., Martin, M. T., Reif, D. M., Houck, K. A., Knudsen, T. B., Rotroff, D. M., Xia, M.H., Sakamuru, S., Huang, R.L., Shinn, P., Austin, C.P., Kavlock, R.J., and D.J. Dix. 2010. Analysis of eight oil spill dispersants using rapid, in vitro tests for endocrine and other biological activity. *Environmental Science & Technology* 44(15): 5979-5985.

Kang, L., Shi, S., Deng, Z., Jin, J., Zhang, F., and H. Huang. 2013. Simulation, visualization and GIS analysis based on globe model for PL19-3 oil spill of Bohai Sea. In: Ulrich, M., Civco, D.L., Schulz, K., Ehlers, M. and K.G. Nikolakopoulos (eds.) *Proceedings of SPIE 8893 Earth Resources and Environmental Remote Sensing/GIS Applications IV, 88931A*. International Society for Optics and Photonics.

Keller, A.A. and K. Clark. 2008. Oil recovery with novel skimmer surfaces under cold climate conditions. In: *Proceedings of the 2008 International Oil Spill Conference, May 4-8, Savannah, Georgia.* American Petroleum Institute. Pp. 667-671.

Khan, F.I., Sadiq, R., and T. Husain. 2002. Risk-based process safety assessment and control measures design for offshore process facilities. *Journal of Hazardous Materials* 94: 1-36.

Kirby, M. F., and R.J. Law. 2010. Accidental spills at sea –Risk, impact, mitigation and the need for co-ordinated post-incident monitoring. *Marine Pollution Bulletin* 60(6): 797-803.

Kulapina, G.M., Markova, O.V., and E.V. Kulapina. 2015. Adapted management techniques for innovative networks. *Review of European Studies* 7(6): DOI: 10.5539/res.v7n6p.1

Laborde, D., Wenke, A., Aarnes, Ø., Rudberg, A., and Ø. Endresen. 2015. Platform for oil spill response gap analysis case Barents and Norwegian Seas. In: *Offshore Mediterranean Conference and Exhibition, 25-27 March, Ravenna, Italy.* Offshore Mediterranean Conference.

Lacoursière, J. P., Dastous, P. A., and S. Lacoursière. 2015. Lac-Mégantic accident: What we learned. Process Safety Progress 34(1): 2-15.

Lamarche, A., MacDonald, S., and S. Grenon. 2013. Development of a Google Earth based contingency planning support system. In: *Proceedings of the 36th AMOP Technical Seminar on Environmental Contamination and Response, Halifax, Canada.* Environment Canada, Ottawa. Pp. 809-817.

Lambert, P. 2003. A literature review of portable fluorescence-based oil-in-water monitors. *Journal of Hazardous Materials* 102(1): 39-55.

Le Floch, S., Dussauze, M., Merlin, F. X., Claireaux, G., Theron, M., Le Maire, P., and A. Nicolas-Kopec. 2014. DISCOBIOL: Assessment of the impact of dispersant use for oil spill response in coastal or estuarine areas. In: *Proceedings of the 2014 International Oil Spill Conference*. American Petroleum Institute. pp. 491-503.

Leifer, I., Clark, R., Jones, C., Holt, B., Svejkovsky, J., and G. Swayze. 2011. Satellite and airborne oil spill remote sensing: State of the art and application to the BP Deepwater Horizon oil spill. In: *Proceedings of the 34th AMOP Technical Seminar on Environmental Contamination and Response, Banff, Canada.* Environment Canada, Ottawa. Pp. 270-295.

Leifer, I., Murray, J., Streett, D., Stough, T., Ramirez, E., and S. Gallegos. 2015. The Federal Oil Spill Team for Emergency Response Remote Sensing, FOSTERRS: Enabling remote sensing technology for marine disaster response. In: Lippitt, C., Stow, D. and L. Coulter (eds.) *Time-Sensitive Remote Sensing*. Springer, New York. pp. 91-111.

Leong, W. F. 2012. Applicability of dynamic risk assessment in the oil and gas industry. In: *SPE Middle East Health, Safety, Security, and Environment Conference and Exhibition*. Society of Petroleum Engineers.pp. 70-76.

Leschine, T. M., Pavia, R., Walker, A. H., Bostrom, A., and K. Starbird. 2015. What-if scenario modeling to support oil spill preparedness and response decision-making. *Human and Ecological Risk Assessment: An International Journal* 21(3): 646-666.

Li, P. 2014. Development of an Integrated Decision Support System for Supporting Offshore Oil Spill Response in Harsh Environments. Doctoral Dissertation, Memorial University of Newfoundland.

Li, P., Chen, B., Li, Z.L., Zheng, X., Wu, H.J., Jing, L., and K. Lee. 2014a. A Monte Carlo simulation based two-stage adaptive resonance theory mapping approach for offshore oil spill vulnerability index classification. *Marine Pollution Bulletin* 86(1-2): 434-442.

Li, P., Chen, B., Zhang, B.Y., Jing, L., and J.S. Zheng. 2012a. A multiple -stage simulation-based mixed integer nonlinear programming approach for supporting offshore oil spill recovery with weathering processes. *Journal of Ocean Technology* 7(4): 87-105.

Li, P., Chen, B., Zhang, B.Y., Jing, L., and J.S. Zheng. 2014b. Monte Carlo simulation -based dynamic mixed integer nonlinear programming for supporting oil recovery and devices allocation during offshore oil spill responses. *Ocean & Coastal Management* 89: 58-70.

Li, P., Chen, B., Li, Z.L. and Jing, L. 2014c. An agent-based simulation-optimization coupling approach for device allocation and operation control in response to offshore oil spills under uncertainty. Proceedings of the International Conference on Marine and Freshwater Environments (iMFE 2014), August 6-8, 2014, St. John's, Canada.

Li, Y., Lan, G. X., Liu, B. X., and D. Chen. 2012b. Dynamic analysis on oil spill in Mexico Bay based on remote sensing and GIS. *Marine Environmental Science* 31(1): 79-82.

Liao, Z., Hannam, P.M., Xia, X. and T. Zhao. 2012b. Integration of multi -technology on oil spill emergency preparedness. *Marine Pollution Bulletin* 64: 2117–2128.

Lieske, D.J., Mahoney, M., Wilhelm, S.I., Weir, L., and P. O'Hara. 2011. Eyes in the sky: A comparative spatial analysis of aerial and satellite surveillance of each coast Canadian oil pollution. In: *Proceedings of the 34th AMOP Technical Seminar on Environmental Contamination and Response, Banff, Canada*. Environment Canada, Ottawa. Pp. 244-258.

Lin, G.H., Chen, X.J. and M. Fukushima. 2009. Solving stochastic mathematical programs with equilibrium constraints via approximation and smoothing implicit programming with penalization. *Mathematical Programming* 116: 343-368.

Linkov, I., Satterstrom, F. K., Kiker, G., Batchelor, C., Bridges, T., and E. Ferguson. 2006. From comparative risk assessment to multi -criteria decision analysis and adaptive management: Recent developments and applications. *Environment International* 32(8): 1072-1093.

Liu, A. X., Lang, Y. H., Xue, L. D., Liao, S. L., and H. Zhou. 2009. Probabilistic ecological risk assessment and source apportionment of polycyclic aromatic hydrocarbons in surface sediments from Yellow Sea. *Bulletin of Environmental Contamination and Toxicology* 83(5): 681-687.

MacPherson, J. 2014. Hiland Crude Pipeline Spills Oil Near Alexander, ND. Huffington. 23 March 2014. http://www.huffingtonpost.com/2014/03/21/hiland-crude-pipeline-spill_n_5008405.html (accessed September 10, 2015)

Margesin, R. and F. Schinner. 2001. Bioremediation (natural attenuation and biostimulation) of diesel -oil-contaminated soil in an alpine glacier skiing area. *Applied and Environmental Microbiology* 67(7): 3127-3133.

Maritime New Zealand. 2006. New Zealand Marine Oil Spill Response Strategy. Maritime New Zealand, New Zealand.

Marthiniussen, R. Vestgard, K., Klepaker, R. A. and N. Storkersen. 2004. HUGIN -AUV concept and operational experiences to date. In: *Ocean '04 - MTS/IEEE Techno-Ocean '04: Bridges across the Oceans - Conference Proceedings, volume 2, , Kobe, Japan, Nov. 2004*. Pg. 846–850.

Martin, J. D., Adams, J., Hollebone, B., King, T., Brown, R. S., and P.V. Hodson. 2014. Chronic toxicity of heavy fuel oils to fish embryos using multiple exposure scenarios. *Environmental Toxicology and Chemistry* 33(3): 677-687.

Martini, N., and R. Patruno. 2005. Oil pollution risk assessment and preparedness in the east Mediterranean. In: *Proceedings of the 2005 International Oil Spill Conference* . American Petroleum Institute. pp. 259-264.

Marty, J., and S. Potter. 2014. Risk Assessment for marine spills in Canadian waters, Phase 1: Oil spills south of the 60th Parallel. In: *Proceedings of the 37th AMOP Technical Seminar on Environmental Contamination and Response, Canmore, Canada* . Environment Canada, Ottawa. Pp. 537-552.

Matheny, K. 2014. Coast Guard: We can't adequately respond to Great Lakes heavy oil spill, Detroit Free Press (September 11, 2014). http://www.freep.com/story/news/local/michigan/2014/09/11/coast-guard-we-cant-adequately-respond-to-great-lakes-heavy-oil-spill/15422415/ (accessed September 18, 2015

Maykuth, A. 2014. 2,550 barrels of crude recovered from oil spill. *Philly.com*. http://articles.philly.com/2014-10-22/business/55284991_1_oil-spill-mid-valley-pipeline-underground-pipeline. (accessed September 10, 2015)

MCA. 2007. The UK SCAT Manual: A Field Guide to the Documentation of Oiled Shorelines in the UK. Maritime & Coastguard Agency, Southampton, UK.

ED_001324_00007647-00443

McGowan, E., Song, L., Marshall, M., and K. River. 2012. The dilbit disaster: Inside the biggest oil spill you've never heard of, Part 1. *Inside Climate News*, June 26, 2012.

McGrath, C. G. G. 2014. Preventing a nationally-significant oil spill on the Grand Banks of Newfoundland. In: *Proceedings of the 2014 International Oil Spill Conference* . American Petroleum Institute. pp. 1226-1238.

Menzie, C.A., Deardorff, T., Booth, P., and T. Wickwire. 2012. Refocusing on nature: Holistic assessment of ecosystem services. *Integrated Environmental Assessment and Management* 8(3): 401-411.

Merchant, R. M., Elmer, S., and N. Lurie. 2011. Integrating social media into emergency -preparedness efforts. *New England Journal of Medicine* 365(4): 289-291.

Metzger, M. J., Rounsevel l, M. D. A., Acosta -Michlik L., Leemans, R. and D. Schröter. 2006. The vulnerability of ecosystem services to land use change. *Agriculture, Ecosystems & Environment* 114(1): 69-85.

Michigan Department of Community Health. 2014. Public Health Assessment: Kal amazoo River/Enbridge Spill: Evaluation of Kalamazoo River Surface Water and Fish after a Crude Oil Release. Michigan Department of Community Health, Lansing.

Minister of Justice. 2001. *Canada Shipping Act, 2001* (2001, c. 26). Published by the Minister of Justice. Ottawa, Canada

Morović, M., and A. Ivanov. 2011. Oil Spill Monitoring in the Croatian Adriatic Waters: needs and possibilities. *Acta Adriat* 52(1): 45-56.

Morris, B. F. and T.R. Loughlin. 1994. Overview of the Exxon Valdez oil spill, 1989 -1992. In: Loughlin, T.R. (ed.) *Marine Mammals and the Exxon Valdez.* Academic Press.

NBC. 2014. Israel Oil Spill: 600,000 -Gallons Lost in Eilat Pipeline Breach. *NBC News* Dec 4 2014. http://www.nbcnews.com/news/world/israel-oil-spill-600-000-gallons-lost-eilat-pipeline-breach-n261386. (accessed September 7, 2015)

NEB (National Energy Board). 2014a. Order XO -E101-003-2014. Orders to Enbridge Pipelines Inc. – Application for the Line 9B Revesrsal and Line 9 Capacity Expansion Project (A59174) . National Energy Board.

NEB. 2014b. Pipeline Performance Measures and Integrity Inspection Information Reporting Form. National Energy Board. http://www.neb-one.gc.ca/sftnvrnmnt/sft/pplnprfrmncmsr/index-eng.html.

NEB. 2015a. Safety. National Energy Board. http://www.neb-one.gc.ca/sftnvrnmnt/sft/index-eng.html. (accessed August 20, 2015).

NEB. 2015b. Enbridge Pipelines Inc. - Line 9B Reversal and Line 9 Capacity Expansion Project (OH -002-2013). National Energy Board. http://www.neb-one.gc.ca/pplctnflng/mjrpp/ln9brvrsl/index-eng.html. (accessed August 20, 2015

NEB. 2015c. Safety and Environmental Performance Da shboard. National Energy Board Updated July 31, 2015. http://www.neb-one.gc.ca/sftnvrnmnt/sft/dshbrd/index-eng.html. (accessed August 2015)

Newbrey, J. L., Paszkowski, C. A., and B.A. Gingras. 2012. Trophic relationships of two species of grebe on a prairie lake based on stable isotope analysis. *Hydrobiologia* 697(1): 73-84.

Nicholls, K., Picou, J. S., and J. Lowman. 2014. Enhancing the utility of community health workers in disaster preparedness, resiliency, and recovery. In: *Proceedings of the 2014 International Oil Spill Conference*. American Petroleum Institute. pp. 170-183.

NRC. 2013. An Ecosystem Services Approach to Assessing the Impacts of the Deepwater Horizon Oil Spill in the Gulf of Mexico. Report by the Committee on the Effects of the Deepwater Horizon Mississippi Canyon-252 Oil Spill on Ecosystem Services in the Gulf of Mexico. Ocean Studies Board, Division on Earth and Life Studies, National Research Council. The Nation al Academies Press, Washington, D.C.

TSB. 2015. Statistical Summary Railway Occurrences 2014. Transportation Safety Board of Canada. Catalogue No. TU1-2E-PDF. June 2015.

30 June 2015

Nuka Research and Planning Group. 2010. Oil Spill Prevention and Response in the US Arctic: Unexamined Risks, Unacceptable Consequences. US Arctic Program, Pew Environment Group.

NYSDEC. 2015. Biomonitoring. Stream Biomonitoring Unit, Division of Water. New York State Department of Environmental Conservation (NYSDEC), Troy, NY . http://www.dec.ny.gov/chemical/23847.html. (accessed June 20, 2015)

OGP (International Association of Oil & Gas Producers). 2011. Process Safety -Recommended Practice on Key Performance Indicators.

OGP. 2011. Process safety - recommended practice on key performance indicators. International Association of Oil & Gas Producers. Report No. 456.

ED_001324_00007647-00444

Øien, K. 2010. Remote operation in environmentally sensitive areas: development of early warning indicators. In: *Proceedings of the 2nd iNTeg-Risk Conference: New Technologies and Emerging Risks - Dealing with multiple and interconnected emerging risks, Stuttgart, Germany.* Steinbeis Editon.

Øien, K., Massaiu, S., Tinmannsvik, R. K., and F. Størseth. 2010. Development of early warning indicators based on Resilience Engineering. In: *PSAM10, International Probabilistic Safety Assessment and Management Conference*. pp. 7-11.

Olita, A., Cucco, A., Simeone, S., Ribotti, A., Fazioli, L., Sorgente, B., and R. Sorgente. 2012. Oil spill hazard and risk assessment for the shorelines of a Mediterranean coastal archipelago. *Ocean & Coastal Management* 57: 44-52.

Ornitz, B. and M. Champ. 2003. Oil Spills First Principles: Prevention and Best Response. Elsevier, Netherlands.

Oskins, C.J., and D. Bradley. 2005. "Extreme" cold weather oil spill response techniques & strategies - ice & snow environments. In: *Proceedings of the 2005 International Oil Spill Conference.* American Petroleum Institute. pp. 521-525.

Owens, E. H. and P.D. Reimer. 2013. Surveying oil on the shoreline. In: Wiens, J.A. (ed.) *Oil in the Environment: Legacies and Lessons of the Exxon Valdez Oil Spill* . Cambridge U niversity Press. Pp. 78-97.

Owens, E.H. and G.A. Sergy. 1994. Field Guide to the Documentation and Description of Oiled Shorelines. Environment Canada, Edmonton, AB, 66 pp.

Owens, E.H. and G.A. Sergy. 2000. The SCAT Manual: A Field Guide to the Documentat ion and Description of Oiled Shorelines. Second Edition. Environment Canada, Edmonton, AB.

Owens, E.H. and G.A. Sergy. 2004. The Arctic SCAT Manual: A Field Guide to the Documentation of Oiled Shorelines in Arctic Environments. Environment Canada, Edmonton, AB.

Pacific Streamkeepers Federation. 2003. The Streamkeepers Program. http://www.pskf.ca/program/program.html.(accessed June 21, 2015)

Paltrinieri, N., Cozzani, V., Øien, K., and T.O. Grøtan. 2012. Prevention of atypical accident scenarios through the use of resilience based early warning indicators. In: Berenger, C., Grall, A. and C. Guedes Soares (eds.) *Advances in Safety, Reliability and Risk Management.* Taylor & Francis, London.

Pan, G., Tang, D ., and Y. Zhang. 2012. Satellite monitoring of phytoplankton in the East Mediterranean Sea after the 2006 Lebanon oil spill. *International Journal of Remote Sensing* 33(23): 7482-7490.

Panzar, J., Reyes, E.A. and J. Mozingo. 2015. Santa Barbara County oil c leanup continues; pipeline may be dug up soon. *Los Angeles Times* May 23, 201.5

Paul, J. H., Hollander, D., Coble, P., Daly, K. L., Murasko, S., English, D., Basso, J., Delaney, J., McDaniel, L., and C.W. Kovach. 2013. Toxicity and mutagenicity of Gulf of M exico waters during and after the Deepwater Horizon oil spill. *Environmental Science & Technology* 47(17): 9651-9659.

Payne, K. C., Jackson, C. D., Aizpurua, C. E., Rojas, O. J., and M.A. Hubbe. 2012. Oil spills abatement: factors affecting oil uptake by cellulosic fibers. *Environmental Science & Technology* 46(14): 7725-7730.

Peltier, H., Jepson, P. D., Dabin, W., Deaville, R., Daniel, P., Van Canneyt, O., and V. Ridoux. 2014. The contribution of stranding data to monitoring and conservation strategies for cetaceans: Developing spatially explicit mortality indicators for common dolphins (*Delphinus delphis*) in the eastern North-Atlantic. *Ecological Indicators* 39: 203-214.

Pilipenko, N., Lerondeau, B., and E. Vial. 2013. Oil spill response in the arctic. In: Proceedings of the 2013 *SPE Arctic and Extreme Environments Technical Conference and Exhibition.* Society of Petroleum Engineers.

Pisano, A. 2011. Development of Oil Spill Detection Techniques f or Satellite Optical Sensors and their Application to Monitor Oil Spill Discharge in the Mediterranean Sea. PhD Thesis. University of Bologna.

Potter, S., Buist, I., Trudel, K, Dickins, D. and E. Owens. 2012. Spill Response in the Arctic Offshore. The American Petroleum Institute and the Joint Industry Programme on Oil Spill Recovery in Ice.

Prendergast, D. P., and P.M. Gschwend. 2014. Assessing the performance and cost of oil spill remediation technologies. *Journal of Cleaner Production* 78: 233-242.

Prince, R. C. 2015. Oil spill dispersants: boon or bane? *Environmental Science & Technology* 49(11): 6376-6384.

ED_001324_00007647-00445

Pula, R., Khan, F.I., Veitch, B., and P.R. Amyotte. 2005. Revised fire consequence models for offshore quantitative risk assessment. *Journal of Loss Prevention in the Process Industries* 18: 443-454.

Radović, J. R., Aeppli, C., Nelson, R. K., Jimenez, N., Reddy, C. M., Bayona, J. M., and J. Albaigés. 2014. Assessment of photochemical processes in marine oil spill fingerprinting. *Marine Pollution Bulletin* 79(1): 268-277.

Ramseur, J., Lattanzio, R., Luther, L., Parfomak, P., and N. Carter. 2014. Oil Sands and the Keystone XL Pipeline: Background and Selected Environmental Issues. Congressional Research Service, Washington, DC.

Reich, D., Balouskus, R., Mc Cay, D.F., Etkin, D.S., Michel, J., and J. Lehto. 2014. An environmental vulnerability model for oil spill risk analyses: Examples from an assessment for the state of Alaska. In: *Proceedings of the 37th AMOP Technical Seminar on Environmental Contamination and Response, Canmore, Canada*. Pp. 65-88.

Reilly, T.I. and R.K. York. 2001. Guidance on Sensory Testing and Monitoring of Seafood for Presence of Petroleum Taint Following an Oil Spill. NOAA Technical Memorandum NOS OR&R 9, Seattle, WA.

Reynolds, K. 2014. Applying international standards in response to oil spills in remote areas. In: *Proceedings of the 2014 International Oil Spill Conference Proceedings* . American Petroleum Institute. pp. 1859-1868.

Rise, I. H. 2014. The Agreement on Cooperation on Marine Oil Pollution Preparedness and Response in the Arctic. Master Thesis. The Arctic University of Norway.

Rost, G., Londong, J., Dietze, S., and G. Osor. 2014. Integrated urban water management –development of an adapted management approach Da rkhan, Kharaa catchment, Mongolia. *Environmental Earth Science* 73: 709–718.

Salt, D., Hughes, E., and S. Gair. 2012. Monitoring and quantifying oil pollution using aerial surveillance - problems and solutions. In: *Proceedings of the 2012 SPE International Co nference on Health, Safety and Environment in Oil and Gas Exploration and Production* . Society of Petroleum Engineers.

Sanyal, J., Zhang, S., Dyer, J., Mercer, A., Amburn, P., and R.J. Moorhead. 2010. Noodles: A tool for visualization of numerical weather model ensemble uncertainty. *IEEE Transactions on Visualization and Computer Graphics* 16(6): 1421-1430.

Shupe, D., Ott, G., and K. Preble. 2014. Evaluating place of refuge risk in the Chesapeake Bay. In: *Proceedings of the 2014 International Oil Spill Confe rence Proceedings.* American Petroleum Institute. pp. 814-824.

Silliman., B. D. 2014. Guidelines to Prepare for Oil Sands Product Spills in Varied Aquatic Environments. In: *Proceedings of the 2014 International Oil Spill Conference* . American Petroleum Institute. pp. 426-433.

Skogdalen, J. E., Utne, I. B., and J.E. Vinnem. 2011. Developing safety indicators for preventing offshore oil and gas deepwater drilling blowouts. *Safety Science* 49(8): 1187-1199.

Song, K. C., Hu, S. P., Yan, Y. H., and J. Li. 2014. Surface defect detection method using saliency linear scanning morphology for silicon steel strip under oil pollution interference. *ISIJ International* 54(11): 2598-2607.

Soomere, T. 2012. A preventative method for minimizing environ mental risks based on the optimization of the location of potentially dangerous activities. In: *Proceedings of the 35th AMOP Technical Seminar on Environmental Contamination and Response, Vancouver, Canada*. Pp. 836-881.

Stephenson, M., Mazzocco, P., and A. St-Amand. 2015. Ecological risk assessment of hypothetical spills of diluted bitumen in rivers. In: *Proceedings of the 38th AMOP Technical Seminar on Environmental Contamination and Response, Vancouver, Canada.*

Struzik, E. 2015. Oil drilling in Arctic Ocean: A push into uncharted waters. *Yale Environment 360*, June 8, 2015.

Sweeten, D.W., Wittman, J., Le Roux, P., and T. Bleier. 2012. Improving oil spill response decision making using a rapidly deployed aerial imaging platform. In: *Proceedings of the 35th AMOP Technical Seminar on Environmental Contamination and Response, Vancouver, Canada* . Pp. 658-666.

Tan, S. H. 2011. In -situ fluorometry and SMART Protocol – The Montara Wellhead Experience. Proceedings of the 2011 International Oil Spill Conference. Ameri can Petroleum Institute. P. Abs141.

ED_001324_00007647-00446

Taylor, E., Moyano, M. and A. Steen. 2014. Upgraded RETOS™: An International Tool to Assess Oil Spill Response Planning and Readiness. In: Proceedings of the 2014 International Oil Spill Conference. American Petroleum Institute. pp. 1353-1363.

The Canadian Press. 2015a. Alberta pipeline spill: Too soon to know what caused Nexen leak, says regulator. *HuffPost Alberta* July 17, 2015. http://www.huffingtonpost.ca/2015/07/17/nexen-pipeline-spills-13_n_7819438.html.

The Canadian Press. 2015b. Regulator orders Nexen Energy to suspend operation of 95 pipelines in Alberta. *CTV News* August 28, 2015. http://www.ctvnews.ca/business/regulator-orders-nexen-energy-to-suspend-operation-of-95-pipelines-in-alberta-1.2538815.

Transport Canada. 2013. A Review of Canada's Ship-Source Oil Spill Preparedness and Response Regime: Setting the Course for the Future. Tanker Safety Panel Secretariat, Ottawa. www.tc.gc.ca/eng/tankersafetyexpertpanel/menu.htm. (accessed June 16 2015).

Transport Canada. 2014. Marine Oil Spill Preparedness and Response Regime. Report to Parliament 2006 – 2011, TP 14539E, Transport Canada, Ottawa, ON.

Transport Canada 2015a. FAQs - Rail Safety and the Transportation of Dangerous Goods by rail. http://www.tc.gc.ca/eng/mediaroom/faq-rail-safety-tsb-7565.html

Transport Canada. 2015b. Joint Industry-Government Guidelines for the Control of Oil Tankers and Bulk Chemical Carriers in Ice Control Zones of Eastern Canada. TP 15163, Transport Canada, Ottawa, ON.

Transport Canada. 2015c. Motor Carriers, Commercial Vehicles and Drivers. https://www.tc.gc.ca/eng/motorvehiclesafety/roadsafety-motor-carriers-commercial-vehicles-drivers-1312.htm

Transport Canada. 2015d. Risk Assessment for Marine Spills in Canadian Waters. http://www.tc.gc.ca/eng/marinesafety/menu-4100.htm#abs. (accessed June 24, 2015)

Transport Canada. 2015e. Tanker Safety and Spill Prevention. http://www.tc.gc.ca/eng/marinesafety/menu-4100.htm. (accessed August 24, 2015)

Turner, M. 2010. Review of Offshore Oil-Spill Prevention and Remediation Requirements and Practices in Newfoundland and Labrador. Department of Natural Resources, Government of Newfoundland and Labrador, Canada.

Tyler-Walters, H., and A. Jackson. 1999. Assessing seabed species and ecosystems sensitivities; rationale and user guide. Report to English Nature, Scottish Natural Heritage and the Department of the Environment Transport, and the Regions from the Marine Life Information Network (MarLIN report 4). Marine Biological Association of the United Kingdom, Plymouth.

US Chemical Safety and Hazard Investigation Board. 2007. Investigation Report, Refinery Explosion and Fire (15 Killed, 180 Injured), BP Texas City, Texas, March 23, 2005. Report No. 2005-04-I-TX, March 2007.

US Coast Guard. 2001. Characteristics of Response Strategies: A Guide for Spill Response Planning in Marine Environments. A joint publication of the American Petroleum Institute, National Oceanic and Atmospheric Administration, US Coast Guard, and US Environmental Protection Agency.

US EPA. 1999. Screening Level Ecological Risk Assessment Protocol (SLERAP) for Hazardous Waste Combustion Facilities. EPA 530-D-99-001A, US Environmental Protection Agency.

US EPA. 2004. Generic Ecological Assessment Endpoints (GEAE) for Ecological Risk Assessment. EPA/630/P-02/004F, US Environmental Protection Agency, Risk Assessment Forum, Washington, DC.

US GAO (US Government Accountability Office). 2012. Oil and Gas: Interior Has Strengthened its Oversight of Subsea Well Containment, but Should Improve its Documentation. Report to Congressional Requesters. GAO-12-244. http://www.gao.gov/assets/590/588961.pdf. (accessed June 15 2015).

US NOAA (National Oceanic and Atmospheric Association). 1990. Excavation and rock washing treatment technology: Net environmental benefit analysis. Hazardous Materials Response Branch, National Oceanic and Atmospheric Administration, Seattle, Washington.

US NOAA. 1994. Alaska Shoreline Countermeasures Manual. Hazardous Materials Response & Assessment Division, National Oceanic and Atmospheric Administration. 99pp

US NOAA. 2013. Shoreline Assessment Manual, 4th Edition. US Dept. of Commerce, Seattle, WA: Emergency Response Division, Office of Response and Restoration, National Oceanic and Atmospheric Administration. 73 pp + appendices.

von Stackelberg, K.E. 2013. Decision analytic strategies for integrating ecosystem services and risk assessment. *Integrated Environmental Assessment and Management* 9(2): 260-268.

Wang, C., Chen, B., Zhang, B., Guo, P. and M. Zhao. 2014. Study of weathering effects on t he distribution of aromatic steroid hydrocarbons in crude oils and oil residues. *Environmental Science: Processes Impacts (Formerly Journal of Environmental Monitoring)* 16(10): 2408-2414.

Wang, C.Y., Chen, B., Zhang, B., He, S. and M. Zhao. 2013. Fingerpri nt and weathering characteristics of crude oils after Dalian oil spill, China. *Marine Pollution Bulletin* 71(1): 64-68

Wang , Y. Z., Yan, Q. B., Huang, J. J., Li, B., and J.L. Gao. 2012. Recovery of lubricating oil for cold strip rolling. Applied Mechanics and Materials 189: 44-47.

Wang , Z., Li, K., Fingas, M., Sigouin, L. and L. Ménard. 2002. Characterization and source identification of hydrocarbons in water samples using multiple analytical techniques. *Journal of Chromatography A* 971: 173-184.

Wang, Z., Hollebone, B. P., Fingas, M., Fieldhouse, B., Sigouin, L., Landriault, M., Smith, P., Noonan, J., Thouin, G. and J.W. Weaver. 2003. Characteristics of Spilled Oils, Fuels, and Petroleum Products: 1. Composition and Properties of Selected Oils. National Exposure Research Laboratory, Office of Research and Development, United States Environmental Protection Agency, Research Triangle Park, North Carolina.

Wang, Z., Yang, C., Yang, Z., Sun, J., Hollebone, B., Brown, C., and M. Landriault. 2011. Forensic fingerprinting and source identification of the 2009 Sarnia (Ontario) oil spill. *Journal of Environmental Monitoring* 13(11): 3004-3017.

Wang, Z.D. and S. Stout. 2006. Oil Spill Environmental Forensics: Fingerprinting and Source Identification, 1st Edition. Academic Press.

Wei, A., Jianwei, L., Yupeng, Z., Haibo, C., Yonggang, W., and S. Tianyun. 2015. R&D of underwear oil spill numerical simulation and 3D visualization system in deepwater area. *Aquatic Procedia* 3: 165-172.

White, H. K., Hsing, P. Y., Cho, W., Shank, T. M., Cordes, E. E., Quattrini, A. M., Nelson, R.K., Camilli, R., Demopoulos, A.W.J., German, C.R., Brooks, J.M., Roberts, H.H., Shedd, W., Reddy, C.M., and C.R. Fisher. 2012. Impact of the Deepwater Horizon oil spill on a deep -water coral community in the Gulf of Mexico. *Proceedings of the National Academy of Sciences* 109(50): 20303-20308.

WHOI. 2010. WHOI Scientists Map and Confirm Origin of Large, Underwater Hydrocarbon Plume in Gulf. Media Relations Office, August 19, 2010. Wo ods Hole Oceanographic Institute. http://www.whoi.edu/page.do?pid=7545&tid=282&cid=79926&. (accessed June 20, 2015)

WSP. 2014a. Risk Assessment for Marine Spills in Canadian Waters. Ph ase 1: Oil Spills South of 60 th Parallel. Final Study Report for Transport Canada. 131-17593-00.

WSP. 2014b. Risk Assessment for Marine Spills in Canadian Waters. Phase 2, Part B: Spills of Oil and Select Hazardous and Noxious Substances (HNS) Transported in Bulk North of the 60 th Parallel North. Final Study Report for Transport Canada. 131-17593-00.

Yan H. 2015. After oil spilled in Yellowstone River, residents told not to drink water. CNN. January 21, 2015. http://edition.cnn.com/2015/01/20/us/yellowstone-river-spill. (accessed September 7, 2015)

Yang, Z., Hollebone, B. P., Wang, Z., Yang, C., Brown, C., and M. Landriault. 2013a. Forensic identification of spilled biodiesel and its blends with petroleum oil based on fingerprinting information. *Journal of Separation Science* 36(11): 1788-1796.

Yang, C., Wang, Z., Liu, Y., Yang, Z., Li, Y., Shah, K., and S. Tian. 2013b. Arom atic steroids in crude oils and petroleum products and their applications in forensic oil spill identification. *Environmental Forensics* 14(4): 278-293.

Yee, D., Dion, F -O., Smith, C., Videla, P., and T. Rodolakis. 2015. Human health risk assessment: acute and chronic biological effects from hypothetical releases of diluted bitumen into sensitive watercourses. In: *Proceedings of the 38h AMOP Technical Semi nar on Environmental Contamination and Response, Vancouver, Canada.*

Yim, U. H., Ha, S. Y., An, J. G., Won, J. H., Han, G. M., Hong, S. H., Kim, M., Jung, J. H. and W.J. Shim. 2011. Fingerprint and weathering characteristics of stranded oils after the Hebei Spirit oil spill. *Journal of Hazardous Materials* 197: 60-69.

Yim, U. H., Kim, M., Ha, S. Y., Kim, S., and W.J. Shim. 2012. Oil spill environmental forensics: the Hebei Spirit oil spill case. *Environmental Science & Technology* 46(12): 6431-6437.

Ying, L., Li, J., and Z. Qingling. 2011. Microwave remote sensing sea surfaces covered in oil. In: *2011 International Conference on Electric Information and Control Engineering* (ICEICE). IEEE. pp. 2319-2322.

You, F., and S. Leyffer. 2011. Mixed -integer dynamic optimization for oil -spill response planning with integration of a dynamic oil weathering model. *AIChE Journal* 57(12): 3555-3564.

ED_001324_00007647-00448

Yu, F., and Y. Yin. 2011. Simulation and 3D visualization of oil spill on the sea. In: *2011 IEEE International Symposium on VR Innovation* (ISVRI). IEEE. pp. 213-216.

Zacharias, M. A. and E.J. Gregr. 2005. Sensitivity and vulnerability in marine environments: an approach to identifying vulnerable marine areas. Conservation Biology 19(1): 86-97.

Zhong, Z., and F. You. 2011. Oil spill response planning with consideration of physicochemical evolution of the oil slick: A multiobjective optimization approach. *Computers & Chemical Engineering* 35(8): 1614-1630.

Zhu, L., Wu, Y. Q., and J. Yin. 2013. Segmentation of marine spill oil SAR image based on Gabor, Krawtchouk Moments and KFCM. *Advanced Materials Research* 760: 1462-1466.

Chapter 8

AER (Alberta Energy Regulator). 2015a. News Release. AER responding to a pipeline failure near Fort McMurray. Alberta Energy Regulator, Calgary, AB. https://www.aer.ca/documents/news-releases/AERNR2015-08.pdf

AER. 2015b. News Release. AER issues Suspension Order to Nexen. Alberta Energy Regulator, Calgary, AB. https://www.aer.ca/documents/news-releases/AERNR2015-13.pdf

Alpine Environmental and EBA Engineering. 2001. Interim Report on the Effects of the Pembina Oil Spill on Fish Inhabiting the Pine River System, Northeastern British Columbia. Submitted to Pembina Pipeline Corporation, Calgary, AB. 260 pp.

Amec. 2001a. Environmental Assessment of the Pembina Pipeline Oil Spill to the Pine River near Chetwynd, BC. Volume 2 – Water Quality. Submitted to District of Chetwynd. 183 pp.

Amec. 2001b. Environmental Assessment of the Pembina Pipeline Oil Spill to the Pine River near Chetwynd, BC. Volume 1 – Sediment Quality. Submitted to District of Chetwynd. 104 pp.

Amec. 2001c. Environmental Assessment of the Pembina Pipeline Oil Spill to the Pine River near Chetwynd, BC. Volume 3 – Fish Surveys. Submitted to District of Chetwynd. 142 pp.

Anderson, A-M. 2005. Wabamun Lake Oil Spill August 2005: Data Report for Water and Sediment Quality in the Pelagic Area of the Lake (August 4-5 to September 15, 2005). Alberta Environment, Edmonton, AB. 99 pp.

Anderson, M.E. 2014. Early review of potential impacts of the *Deepwater Horizon* oil spill on Gulf of Mexico wetlands and their associated fisheries. In: Alford, J. B., Peterson, M. S. and C. C. Green (eds.) *Impacts of Oil Spill Disasters on Marine Habitats and Fisheries in North America.* CRC Press, Boca Raton, FL. Pp 97-112.

AOSRT 2014. Environmental Impacts of Arctic Oil Spills and Arctic Spill Response Technologies - Literature Review and Recommendations. December 2014. Arctic Oil Spill Response Technology Joint Industry Programme. http://neba.iogp.org/assets/files/Environmental%20Impacts%20of%20Arctic%20Oil%20Spills%20-%20report.pdf

Atlas, R. and T. Hazen. 2011. Oil biodegradation and bioremediation: A tale of the two worst spills in U.S. history. *Environmental Science & Technology* 45: 6709–6715.

Baccante, N. 2000. Preliminary report on fish mortalities in the Pine River, near Chetwynd, B.C., following an oil spill from Pembina's pipeline on August 1, 2000. BC Ministry of Environment, Fort St. John, BC. 20 pp.

Ballachey, B.E., Bodkin, J.L., Esler, D. and S.D. Ricer. 2014. Lessons from the 1989 Exxon Valdez oil spill: A biological perspective. In: Alford, J. B., Peterson, M. S. and C. C. Green (eds.) *Impacts of Oil Spill Disasters on Marine Habitats and Fisheries in North America.* CRC Press, Boca Raton, FL. Pp. 181-197.

Bejarano, A.C., Michel, J., and Williams, L. 2012. Net Environmental Benefit Analysis (NEBA) Relative Risk Ranking Conceptual Design, Kalamazoo River System, Enbridge Line 6B Release. August 8, 2012; document and appendixes; AR-0963. Cited in Fitzpatrick, F.A., Boufadel, M.C., Johnson, R., Lee, K., Graan, T.P., Bejarano, A.C., Zhu, Z., Waterman, D., Capone, D.M., Hayter, E., Hamilton, S.K., Dekker, T., Garcia, M.H. and J.S. Hassan. 2015. Oil-Particle Interactions and Submergence from Crude Oil Spills in Marine and Freshwater Environments – Review of the Science and Future Science Needs. U.S. Geological Survey. Open File Report 2015-1076.

ED_001324_00007647-00449

Biber, P. D., Wu, W., Peterson, M.S., Liu, Z. and L. Pham. 2014. Oil contamination in Mississippi saltmarsh habitats and the impacts to *Spartina alterniflora* photosynthesis. In: Alford, J. B., Peterson, M. S. and C. C. Green (eds.) *Impacts of Oil Spill Disasters on Marine Habitats and Fisheries in North America.* CRC Press, Boca Raton, FL. Pp. 133-171.

Birtwell, I.K. 2003. Comments on the Effects of Crude Oil on Fish and Their Habitat in the Pine River, British Columbia. Fisheries and Oceans Canada, West Vancouver, BC. 25 pp.

Birtwell, I.K. 2008b. Comments on the Effects of Oil Spillage on Fish and Their Habitat – Lake Wabamun, Alberta. Fisheries and Oceans Canada, West Vancouver, BC. 198 pp.

Boehm, P.D., M.S. Steinhauer, D.R. Green, B. Fowler, B. Humphrey, D.I Feist and W.J. Cretney. 1987. Comparative fate of chemically dispersed and beached crude oil in subtidal sediments of the Arctic nearshore. Arctic 40 (Supp, 1): 133-148.

Brandvik, P. J, Johansen, O., Leirvik, F., Farooq, U. and P. S. Daling. 2013. Droplet breakup in sub-surface oil releases—Part 1: Experimental study of droplet breakup and effectiveness of dispersant injection. *Marine Pollution Bulletin* 73: 319-326.

Brannon, E.L., Collins, K., Cronin, M.A., Moulton, L.L., Maki, A.L. and K.R. Parker. 2012. Review of the *Exxon Valdez* spill effects on pink salmon in Prince William Sound, Alaska. *Reviews in Fisheries Science* 20: 20-60.

Boufadel, M.C., Abdollahi-Nasab, A., Geng, X., Galt, J. and J. Torlapati. 2014. Simulation of the landfall of the Deepwater Horizon oil on the shorelines of the Gulf of Mexico. *Environmental Science & Technology* 48: 9496-9505.

Britannica. 2015. Deepwater Horizon Oil Spill of 2010. http://www.britannica.com/event/Deepwater-Horizon-oil-spill-of-2010

Brodersen, C., J. Short, L. Holland, M. Carls, J. Pella, M. Larsen, and S. Rice. 1997. Evaluation of oil removal from beaches 8 years after the *Exxon Valdez* oil spill. pp 325-336. In: Proceedings of the 22nd Arctic & Marine Oilspill Program (AMOP) Technical Seminar, Environment Canada, Ottawa, ON, Canada.

Brown-Peterson, N.J., Brewton, R.A., Griffitt, R.J. and R.S. Fulford. 2014. Impacts of the Deepwater Horizon oil spill on the reproductive biology of Spotted Seatrout (*Cynoscion nebulosus*). In: Alford, J. B., Peterson, M. S. and C. C. Green (eds.) *Impacts of Oil Spill Disasters on Marine Habitats and Fisheries in North America.* CRC Press, Boca Raton, FL.pp. 237-251.

Bunch, J.N. 1987. Effects of petroleum releases on bacterial numbers and microheterotrophic activity in the water and sediment of an arctic marine ecosystem. Arctic 40 (Supp. 1): 172-183.

Camilli, R., Reddy, C. M., Yoerger, D.R., Van Mooy, B.A.S., Jakuba, M.V., Kinsey, J.C., McIntyre, C.P., Sylva, S.P. and J. V. Maloney. 2010. Tracking hydrocarbon plume transport and biodegradation at Deepwater Horizon. *Science* 330: 201-204.

Campo, P., Venosa, A.D. and M. T. Suidan. 2013. Biodegradability of Corexit 9500 and Dispersed South Louisiana Crude Oil at 5 and 25 °C. *Environmental Science & Technology* 47(4): 1960-1967.

CBC. 2006. Alberta establishes new disaster response agency. CBC News October 17, 2006. http://www.cbc.ca/news/canada/edmonton/alberta-establishes-new-disaster-response-agency-1.582879.

CBC. 2008. Alberta band settle Wabamun oil-spill lawsuit. CBC News September 12, 2008. http://www.cbc.ca/news/canada/calgary/alberta-band-settles-wabamun-oil-spill-lawsuit-1.767873

CCG (Canadian Coast Guard). 2015a. Independent Review of the *M/V Marathassa* Fuel Oil Spill Environmental Response Operation. Report presented to Commissioner Jody Thomas, Canadian Coast Guard. July 19, 2015. http://www.ccg-gcc.gc.ca/independent-review-Marathassa-oil-spill-ER-operation

CCG. 2015b. *Marathassa* Spill Operational Update – April 23, 2015. Canadian Coast Guard. http://www.dfo-mpo.gc.ca/media/infocus-alaune/2015/Marathassa/index-eng.htm

Chanton, J., Zhao, T., Rosenheim, B.E., Joye, S., Bosman, S., Brunner, C., Yeager, K.M., Diercks, A.R. and D. Hollander. 2015. Using natural abundance radiocarbon to trace the flux of petrocarbon to the seafloor following the Deepwater Horizon oil spill. *Environmental Science & Technology* 49: 847-854

ED_001324_00007647-00450

Cretney, W.J., D.R. Green, B.R. Fowler, B. Humphrey, D.L. Feist and P.D. Boehm. 1987a. Hydrocarbon biogeochemical setting of the Baffin Island oil spill experimental sites. I: Sediments. Arctic 40 (Supp. 1): 51-65.

Cretney, W.J. 1987b. Hydrocarbon biogeochemical setting of the Baffin Island oil spill experimental sites. II: Water Arctic 40 (Supp. 1): 66-70.

Cretney, W.J., D.R. Green, B.R. Fowler, B. Humphrey, F.R. Engelhardt, R.J. Nordstrom, M. Simon, D.L. Feist and P.D. Boehm. 1987a. Hyddrocarbon biogeochemical setting of the Baffin Island oil spill experimental sites. III: Biota. Arctic 40 (Supp. 1): 71-79.

Cross, W.E., and C.M. Martin. 1983. *In situ* studies of effects of oil and chemically treated oil on primary productivity of ice algae and on under-ice meiofanual and macrofaunal communities. Special studies – 1983 study results. Baffin Island Oil Spill Project Working Report 82-7. Environmental Protection Service, Environment Canada. 103 pp.

Cross, W.E. and D.H. Thompson. 1987. Effects of experimental releases of oil and dispersed oil on Arctic nearshore macrobenthos. I. Infauna. Arctic 40 (Supp. 1): 184-200.

Cross, W.E., R.T. Wilce and M.F. Fabijan. 1987b. Effects of experimental releases of oil and dispersed oil on Arctic nearshore macrobenthos. III; Macroalgae. Arctic 40 (Supp. 1): 211-219.

Cross, W.E., C.M. Martin and D.H. Thomson. 1987b. Effects of experimental releases of oil and dispersed oil on Arctic nearshore macrobenthos. II; Epibenthos. Arctic 40 (Supp. 1): 201-210.

DeBruyn, A.M.H., Wernick, B.G., Stefura, C., McDonald, B.G., Rudolph, B-L., Patterson, L. and P.M. Chapman. 2007. In situ experimental assessment of Lake whitefish development following a freshwater oil spill. *Environmental Science and Technology* 41: 6983-6989.

Dickins, D.F. and J.A. Hellebust. 1981. Return to Balaena Bay: Long-term effect of a large scale crude oil spill under Arctic sea ice. Gylf Canada Resources Inc., Calgary, AB.

Dickins, D. 2011. Behavior of oil spills in ice and implications for Arctic spill response. Offshore Technology Conference. OTC 22126. Houston, Texas, February 709, 2011.

Di Toro, D.M., McGrath, J.A and D.J. Hansen. 2000. Technical basis for narcotic chemicals and polycyclic aromatic hydrocarbon criteria. I. Water and tissue. *Environmental Toxicology and Chemistry* 19: 1951-1970.

Di Toro, D.M. and J.A. McGrath. 2000. Technical basis for narcotic chemicals and polycyclic aromatic hydrocarbon criteria. II. Mixtures and sediments. *Environmental Toxicology and Chemistry* 19:1971-1982.

DNV (Det Norske Veritas). 2011. Final Report Assessment of the Risk of Pollution from Marine Oil Spills in Australian Ports and Waters. Report to Australian Maritime Safety Authority. Report No. PP002916, Rev. 5. Cited in WSP and SL Ross. 2014a. Risk Assessment for Marine Spill in Canadian Waters. Phase 1: Oil Spills South of 60th Parallel. Final Study Report prepared for Transport Canada. http://wcel.org/sites/default/files/file-downloads/131-17593-00_ERA_Oil-Spill-South_150116_pp1-124.pdf.

Donahue, W.F., Allen, E.W., and D.W. Schindler. 2006. Impacts of coal-fired power plants on trace metals and polycyclic aromatic hydrocarbons (PAHs) in lake sediments in central Alberta, Canada. *Journal of Paleolimnology* 35: 111-128.

Eimhjellen,K. and K. Josefson. 1984. Microcbilogy 2: Biodegradation of stranded oil – 1982 study results. Baffin Island Oil Spill Project Worksing Report 83-6. Environmental Proection Service, Environment Canada, Ottawa. 58 pp.

Energy East. 2014a. Accidents and Malfunctions. Volume 6 of the Environmental Impact Assessment. Section 3: Crude oil characteristics, environmental fate, transport and effects. Energy East Pipeline Project. 48 pp. Available online from the National Energy Board at https://docs.neb-one.gc.ca/ll-eng/llisapi.dll/fetch/2000/90464/90552/2432218/2540913/2543426/2543068/A63989-4_ESA_V6_S3_OilFateTranspEffects_-_A4E1F4.pdf?nodeid=2543560&vernum=-2

Energy East. 2014b. Accidents and Malfunctions. Volume 6 of the Environmental Impact Assessment. Section 4: Sites of interest. 112 pp. Available on-line from the National Energy Board at https://docs.neb-one.gc.ca/ll-

eng/llisapi.dll/fetch/2000/90464/90552/2432218/2540913/2543426/2543068/A63989-5_ESA_V6_S4_SitesOfInterest_-_A4E1F5.pdf?nodeid=2543268&vernum=-2

EPA (US Environmental Protection Agency) 1998. Guidelines for Ecological Risk Assessment. EPA/630/R-95/002F.

EPA. 2012. Response Technologies for Oil Sands Products: Enbridge Oil Spill Case Study. Kalamazoo River, Michigan. Presentation. http://www.southportland.org/files/8813/9509/2248/Water_Quaility_Impacts_-_Response_Technologies_for_Oil_Sands_Products__Enbridge_Oil_Spill_Case_Study_Kalamazoo.pdf.

Etkin, D.S. 2015. New risks from crude-by-rail transportation. In: *Proceedings of the Thirty-Eight AMOP Technical Seminar, Environment Canada, Ottawa, ON.* pp 900-923.

Evans, C.L. 2008. An Evaluation of the Harmful Alteration, Disruption and Destruction (HADD) of Fish Habitat Following the Bunker "C" And Pole Treating Oil Release on Wabamun Lake, Alberta. Report submitted to Public Prosecution Service of Canada, Alberta Regional Office, Fisheries and Oceans Canada, Prairies Area Operations, Central and Arctic Region.

EVOSTC. 2015. Status of Injured Resources and Services. Exxon Valdez Oil Spill Trustee Council. http://www.evostc.state.ak.us/index.cfm?FA=status.injured

Federal Interagency Solutions Group. 2010. Oil Budget Calculator: Deepwater Horizon. http://www.restorethegulf.gov/sites/default/files/documents/pdf/OilBudgetCalc_Full_HQ-Print_111110.pdf

Fitzpatrick, F.A. 2014. Effects of Diluted Bitumen in the Environment: Introductory Remarks. Presentation. NAS/NRC 1st Committee Meeting. December 3, 2014. http://nas-sites.org/dilbit/files/2014/12/5_Fitzpatrick_USGS.pdf.

Fitzpatrick, F.A., Boufadel, M.C., Johnson, R., Lee, K., Graan, T.P., Bejarano, A.C., Zhu, Z., Waterman, D., Capone, D.M., Hayter, E., Hamilton, S.K., Dekker, T., Garcia, M.H. and J.S. Hassan. 2015. Oil-Particle Interactions and Submergence from Crude Oil Spills in Marine and Freshwater Environments – Review of the Science and Future Science Needs. U.S. Geological Survey. Open File Report 2015-1076.

Fodrie, F.J. and K.L. Heck Jr. 2011. Response of coastal fishes to the Gulf of Mexico oil disaster. *PLoS ONE* 6: e21609. doi:10.1371/journal.pone.0021609. Cited in Mendelssohn, I.A., Anderson, G.L., Baltz, D.M., Caffey, R.H., Carman, K.R., Fleeger, J.W., Joye, S.B., Lin, Q., Maltby, E., Overton, E.B. and L.P. Rozas. 2012. Oil impacts on coastal wetlands; implications for the Mississippi River Delta ecosystem after the *Deepwater Horizon* oil spill. *BioScience* 62: 562-574.

Foght, J. 2006. Potential for Biodegradation of Sub-Littoral Residual Oil by Naturally Occurring Microorganisms Following the Lake Wabamun Train Derailment. Submitted to Alberta Environment and Environment Canada. Dept. Biological Sciences, University of Alberta.

Fulford, R.S., Griffit, R.J., Brown-Peterson, N.J., Perry, H. and G. Sanchez-Rubio. 2014. Impacts of the *Deepwater Horizon* oil spill on blue crab, *Callinectes sapidus*, larval settlement in Mississippi. In: Alford, J. B., Peterson, M. S. and C. C. Green (eds.) *Impacts of Oil Spill Disasters on Marine Habitats and Fisheries in North America.* CRC Press, Boca Raton, FL. Pp. 253-264.

Garcia-Pineda, O., I. MacDonald, C. Hu, J. Svejkovsky, M. Hess, D. Dukhovskoy, and S.L. Morey. 2013. Detection of floating oil anomalies from the Deepwater Horizon oil spill with synthetic aperture radar. *Oceanography* 26(2):124–137, http://dx.doi.org/10.5670/oceanog.2013.38.

GENIVAR. 2013. Évaluation Environnementale Stratégique sur la Mise en Valeur des Hydrocarbures dans les Bassins d'Anticosti, de Madeleine et de la Baie des Chaleurs. Rapport de GENIVAR au Ministère des Ressources Naturelles. Cited in WSP and SL Ross. 2014a. Risk Assessment for Marine Spill in Canadian Waters. Phase 1: Oil Spills South of 60th Parallel. Final Study Report prepared for Transport Canada. http://wcel.org/sites/default/files/file-downloads/131-17593-00_ERA_Oil-Spill-South_150116_pp1-124.pdf.

George-Ares, A. and J. R. Clark. 2000. Aquatic toxicity of two Corexit dispersants. *Chemosphere* 40: 897-906.

Gilde, K., and J. Pinckney. 2012. Sublethal effects of crude oil on the community structure of estuarine phytoplankton. *Estuaries and Coasts* 35: 853-861.

Goldberg, H. 2011. Pine River 2011 Fisheries Update: Status of Recovery Post-2000 Pipeline Rupture. Report Submitted to Enbridge, c/o Paul Anderson, Director of Environment Northern Gateway Project. Response to Haisla Nation IR 1.52. National Energy Board Northern Gateway Application.

Goldsmith, B.J., Waikem, T.K. and T. Franey. 2014. Environmental damage liability regimes concerning oil spills – A global review and comparison. In: *Proceedings of the 2014 International Oil Spill Conference*. http://ioscproceedings.org/doi/pdf/10.7901/2169-3358-2014.1.2172

Golder Associates Ltd. 2007 Lakewater and Aquatics Long-Term Monitoring –2005-2006 Interpretive Report, Wabamun Lake Derailment Site. CanadianNational Railway Company PIN 2401257 Wabamun, Alberta. Report prepared for CN Railway Co. by Golder Associates Ltd., North Vancouver, B.C. Cited in Birtwell, I.K. 2003. Comments on the Effects of Crude Oil on Fish and Their Habitat in the Pine River, British Columbia. Fisheries and Oceans Canada, West Vancouver, BC.

Gordon, D.C.Jr., Dale, J.and P.D. Keizer. 1978. Importance of sediment working by the deposit-feeding polychaete *Arenicola marina* on the weathering rate of sediment-bound oil. *Journal of the Fisheries Research Board of Canada* 35(5): 591-603.

Government of Alberta. 2015. Alberta Guide to Sportfishing Regulations. http://albertaregulations.ca/fishingregs-pdfs-2015.html

Graham, W. M., Condon, R. H. Carmichael, R. H., D'Ambra, I., Patterson, H.K. Linn, L.J. and F. J. Hernandez. 2010. Oil carbon entered the coastal planktonic food web during the Deepwater Horizon oil spill. *Environmental Research Letters* 5: 045301 (6p).

Green, J., Postlewaite, L. and P. Anderson. 2015. Context for assessing effects on the biophysical and human environment – the pipeline ecological and human health risk assessment for the Northern Gateway Pipeline Project. In: *Proceedings of the Thirty-Eight AMOP Technical Seminar*. Environment Canada, Ottawa, ON, pp 526-535.

Hamilton, S. 2015. Presentation to the Royal Society Expert Panel on the Behaviour and Environmental Impacts of Crude Oil Released in Aqueous Environments. April, 2015, Halifax, NS.

Harwell, M.A. and J.H. Gentile. 2006. Ecological significance of residual exposures and effects from the *Exxon Valdez* oil spill. *Integrated Environmental Assessment and Management* 2: 204-246.

Hauge, K.H., Blanchard, A., Anderson, G. Boland, R. Grøsvik, B.E. Howell, D. Meier, S. Olsen, E. and F. Vikebø. 2014. Inadequate risk assessments –A study on worst-case scenarios related to petroleum exploitation in the Lofoten area. *Marine Policy* 44: 82-89

Hodson, P.V. 2008. Report on the Toxicity of Oil to Fish. Submitted to Fisheries and Oceans Canada. DFO Contract # F2471-080006.

Hodson. P.V. and J. Martin. 2012. Review of Technical Data Report. Ecological and human health risk assessment for pipeline spills: Enbridge Northern Gateway Project. Prepared for the Haisla Nation Council, Kitimaat Village, BC.

Hodson, P.V., Shaw, W., Abudulai, N. and R.S. Brown. 2007. Were Fish in L. Wabamun Exposed to Spilled Oil After the Train Derailment in 2005? Platform presentation, 28th Arctic and Marine Oil spill Program (AMOP) Technical Seminar, Edmonton, AL, June 4 7, 2007.

Hollebone, B.P. 2008. Physical and Chemical Fate and Behaviour of Oil Spilled in Wabamun Lake, August 2005. Environmental Science and Technology Centre, Science and Technology Branch, Environment Canada. 45 pp.

Hollebone, B.P., Fieldhouse, B., Sergey, G., Lamter, P., Wang, Z., Yang, C. and M. Landriaut. 2011. The behaviour of heavy oil in fresh water lakes. In: *Proceedings of the 34th Arctic and Marine Oilspill Program (AMOP) Technical Seminar on Environmental Contamination and Response.* Environment Canada, Ottawa. Pp. 668-709.

Horn, M. and D. French-McCay. 2015. Trajectory and fate modeling with acute effects assessment of hypothetical spills of diluted bitumen into rivers. In: *Proceedings of the Thirty-Eight AMOP Technical Seminar*. Environment Canada, Ottawa, ON, pp 549-581.

Humphrey, B., P.D. Boehm, M.C. Hamilton and R.J. Nordstrom. 1987a. The fate of chemically dispersed and untreated crude oil in Arctic benthic biota. Arctic 40 (Supp. 1): 149-161.

Humphrey, B., D.R. Green, B.r. Fowler, D. Hope and P.D. Boehm. 1987b. The fate of oil in the water column following experimental oil spills in the Arctic marine nearshore. Arctic 40 (Supp. 1): 124-132.

Iverson, S. A. and D. Esler. 2010. Harlequin duck population injury and recovery dynamics following the 1989 *Exxon Valdez* oil spill. *Ecological Applications* 20:1993–2006. Cited in Ballachey, B.E., Bodkin, J.L., Esler, D. and S.D. Ricer. 2014. Lessons from the 1989 Exxon Valdez oil spill: A biological perspective. In: Alford, J. B., Peterson, M. S. and C. C. Green (eds.) *Impacts of Oil Spill Disasters on Marine Habitats and Fisheries in North America.* CRC Press, Boca Raton, FL. Pp. 181-197.

Johansen, Ø, Brandvik, P. J. and U. Farooq. 2013. Droplet breakup in subsea oil releases—Part 2: Predictions of droplet size distributions with and without injection of chemical dispersants. *Marine Pollution Bulletin* 73:327-335.

Kane, L. 2015. Vancouver fuel spill underscores gap in research after federal cuts: aquarium. *Global and Mail* April 16, 2015. http://www.theglobeandmail.com/news/british-columbia/vancouver-fuel-spill-underscores-gap-in-research-after-federal-cuts-aquarium/article23999926/

Kinder Morgan. 2013. Trans Mountain Pipeline (ULC) Transmountain Expansion Project. Volume 7 – Risk Assessment and Management of Pipeline and Facility Spills. + Appendices B 18-2, B18-3, B18-4, B18-5, B18-6, B18-7, B18-8, B18-12. Submitted to the National Energy Board. https://docs.neb-one.gc.ca/ll-eng/llisapi.dll?func=ll&objId=2393783&objAction=browse&viewType=1.

Kinder Morgan. 2014. Trans Mountain Pipeline ULC Trans Mountain Expansion Project. NEB Hearing Order H-001-2014. Responses to Information Request from Village of Belcarra. 2007 Oil Spill Event: 'Post Mortem', 'Lessons Learned' & 'Corrective Actions'. Trans Mountain Response to Village of Belcarra IR No.2. http://www.belcarra.ca/reports/TMEP_Response_to_Belcarra_Information_Request_No.2.pdf.

Kostka, J.E., Prakash, O., Overholt, W.A., Green, S.J., Freyer, G., Canion, A., Delgardio, J., Norton, N., Hazen, T.C. and M. Huettel. 2011. Hydrocarbon-degrading bacteria and the bacterial community response in Gulf of Mexico beach sands impacted by the Deepwater Horizon oil spill. *Applied and Environmental Microbiology* 77: 7962-7974.

Kujawinski, E.B., Kido Soule, M.C., Valentine, D.L., Boysen, A.K., Longnecker, K. and M.C. Redmond. 2011. Fate of dispersants associated with the Deepwater Horizon spill. *Environmental Science and Technology* 45: 1298-1306.

Lake Wabamun Residents Committee. 2007. The Lake Wabamun Disaster: A Catalyst for Change. Report to the Railway Safety Act Review Advisory Panel. August 31, 2007. 12 pp.

Landis, W.G. 2007. The *Exxon Valdez* oil spill revisited and the dangers of normative science. *Integrated Environmental Assessment and Management* 3: 439-441.

Lee, K., Wohlgeschaggen, G.D., Tremblay, G.H., Vandermeulen, J.H., Mossman, D.C., Doe, K.G., Jackman, P.M., Wilson, J.E.H., Prince, R.C., Garrett, R.M. and C.E. Haith. 1999. Natural recovery reduces impact of the 1970 *Arrow* oil spill. In: *Proceedings of the 1999 International Oil Spill Conference*. Pp. 1075-1078.

Lee, K., Prince, R.C., Greer, C.W., Doe, K.G., Wilson, J.E.H., Cobanli, S.E., Wohlgeschaffen, G.D., Alroumi, D., King, T. and G.H. Tremblay. 2003. Composition and toxicity of residual Bunker C fuel oil in intertidal sediments after 30 years. *Spill Science and Technology Bulletin* 8: 187-199.

Lee,K. M. Boudreau, J. Bugden, L. Burridge, S.E. Coblani, S. Courtenay, S. Grenon, B. Hollebone, P. Kepkay, Z. Li, M. Lyons, H. Niu, T.L. King, S. MacDonald, E.C. McIntyre, B. Robinson, S.A. Ryan and G. Wohlgeschaffen. 2011. State of knowledge review of fate and effect of oil in the Arctic marine environment. National Energy Board of Canada. 259 p. Available: https://docs.neb-one.gc.ca/ll-eng/llisapi.dll/fetch/2000/90463/621169/700096/704342/A2A8R2_-_NEB_State_of_Knowledge_Review_of_Fate_and_Effect_of_Oil_DFO_COOGER.pdf?_gc_lang=en&nodeid=704343&vernum=0

Lee, K., Nedwed, T., Prince, R.C. and D. Palandro. 2013. Laboratory tests on the biodegradation of chemically dispersed oil should consider the rapid dilution that occurs at sea. *Marine Pollution Bulletin* 73: 314–318.

La Peyre, J, S. Casa and S. Miles. 2014. Oyster responses to the Deepwater Horizon oil spill across coastal Louisisana: examining oyster health and hydrocarbon bioaccumulation. In: Alford, J. B., Peterson, M. S. and C. C. Green (eds.) *Impacts of Oil Spill Disasters on Marine Habitats and Fisheries in North America.* CRC Press, Boca Raton, FL. Pp. 269-288

Lemly, A.D. 1993. Metabolic stress during winter increases the toxicity of selenium to fish. *Aquatic Toxicology* 27: 133-158. Cited in Birtwell, I.K. 2003. Comments on the Effects of Crude Oil on Fish and Their Habitat in the Pine River, British Columbia. Fisheries and Oceans Canada, West Vancouver, BC.

Levy, E.M. 1972. Evidence for the recovery of the waters off the east coast of Nova Scotia from the effects of a major oil spill. *Water, Air and Soil Pollution* 1: 144-148.

Lilwal, S. and E. Fitzpatrick. 2015. Wabamun Lake oil spill: A decade later, disaster still fresh in resident's minds. CBC News August 3, 2015. http://www.cbc.ca/news/canada/edmonton/wabamun-lake-oil-spill-a-decade-later-disaster-still-fresh-in-residents-minds-1.3177556

Maritime Museum of the Atlantic. 2015. Marine Heritage Database. "On the Rocks: Find a Wreck". *Arrow* – 1970. Online. Arrow Shipwreck - Maritime Museum of the Atlantic fact sheet.htm http://novascotia.ca/museum/wrecks/wrecks/shipwrecks.asp?ID=468

McCleneghan, K. 2008. Analysis of Rsponse to the Lake Wabamun Oil Spill. Prepared for Fisheries and Oceans Canada, Edmonton, Alberta. 101 pp.

McDonald, B.G., deBruyn, A. M.H., Wernick, B.G., Patterson, L., Pilerin, N. and P.M. Chapman. 2007. Design and application of a transparent and scalable weight-of-evidence framework: an example from Wabamun Lake, Alberta, Canada. *Integrated Environmental Assessment and Management* 3: 476-483.

McKenzie, J.F., Schieble, C., Smith, P.W. and M.T. O'Connell. 2014. Occurrence of lemon sharks (*Negaprion brevirostris*) at Chandeleur Islands, Louisiana, before and after the 2010 *Deepwater Horizon* disaster. In: Alford, J. B., Peterson, M. S. and C. C. Green (eds.) *Impacts of Oil Spill Disasters on Marine Habitats and Fisheries in North America.* CRC Press, Boca Raton, FL. Pp. 295-309.

Mendelssohn, I.A., Anderson, G.L., Baltz, D.M., Caffey, R.H., Carman, K.R., Fleeger, J.W., Joye, S.B., Lin, Q., Maltby, E., Overton, E.B. and L.P. Rozas. 2012. Oil impacts on coastal wetlands; implications for the Mississippi River Delta ecosystem after the *Deepwater Horizon* oil spill. *BioScience* 62: 562-574.

Millsap, S., Williams, L., Haas, J., Hanshue, S., Wesley, J., Taft, W., Walterhouse, M., Winter, J., Williamson, R.T., Beltman, D., Ebbets, A., Ritter, K., Tilleitt, D.E., Papoiulias, D. Nicks, D. and P. Badra. 2012. Assessing Natural Resource Impacts from the Enbridge Pipeline Spill into the Kalamazoo River. Presentation. https://crrc.unh.edu/sites/crrc.unh.edu/files/media/docs/Workshops/oil_sands/Winter_naturalresourceimpacts.pdf

NORCOR Engineering & Research Ltd. 1975. The interaction of crude oil with Arctic sea ice. Beaufort Sea Project Technical Report No. 27. Canadian Department of Environment, Victoria, BC. 201 pp.

NRC (National Research Council). 2013. An Ecosystem Services Approach to Assessing the Impacts of the Deepwater Horizon Oil Spill in the Gulf of Mexico. National Research Council of the National Academies. National Academies Press, Washington, DC. http://www.nap.edu/catalog.php?record_id=18387.

Neff JM, Owens EH, Stoker SW, McCormick DM. 1995. Shoreline oiling conditions in Prince William Sound following the Exxon Valdez oil spill. In: Wells PG, Butler JN, Hughes JS, editors. Exxon Valdez oil spill: Fate and effects in Alaskan waters. Philadelphia (PA): American Society for Testing and Materials. ASTM STP 1219. p 312–345.

Noble, S. 2012. Two years after the Enbridge pipeline rupture. EPA Situation Report. Presentation. Michigan Department of Environmental Quality. Enbridge Response Unit. http://www.mi-wea.org/docs/Two%20Year%20after%20the%20Enbridge%20Pipeline%20Rupture%20MWEA.pdf

North Saskatchewan Watershed Alliance. 2015. Riparian Health Assessment of Wabamun Lake: An Aerial Assessment Using an Unmanned Air Vehicle (UAV). Prepared for the Wabamum Watershed Management Council, Edmonton, AB. https://www.nswa.ab.ca/sites/default/files/Riparian%20Health%20Assessment%20Wabamun.pdf.

NTSB (National Transportation Safety Board). 1990. Marine Accident Report: Grounding of the U.S. tankship *Exxon Valdez* on Bligh Reef, Prince William Sound near Valdez, Alaska. March 24, 1989. U.S. National Transportation Safety Board. NSB/MAR-90/04. PB90-916405.

http://docs.lib.noaa.gov/noaa_documents/NOAA_related_docs/oil_spills/marine_accident_report_1990.pdf

NTSB. 2012. Accident Report: Enbridge Incorporated Hazardous Liquid Pipeline Rupture and Release, Marchall, Michigan. July 25, 2010. U.S. National Transportation Safety Board. NTSB/PAR-12/01. Washington, DC. http://www.ntsb.gov/investigations/AccidentReports/Reports/PAR1201.pdf.

Owens, E.H. 1978. Mechanical dispersion of oil stranded in the littoral zone. *Journal of the Fisheries Research Board of Canada* 35: 563-572.

Owens, E.H., J.R. Harper, W. Robson and P.D. Boehm. 1987. Fate and persistence of crude oil stranded on a sheltered beach, Arctic 40 (Supp. 1): 109-123.

Owens EH. 1991. Shoreline conditions following the *Exxon Valdez* spill as of fall 1990. In: Proceedings of the 14th Arctic and Marine Oilspill Program Technical Seminar; 1991 June 12–14; Vancouver, BC. Ottawa (ON): Environment Canada. p 576–606.

Owens, E.H., G.A. Sergy, B.E. McGuire and B. Humphrey. 1993. The 1970 Arrow soil spill – What remains on the shoreline 22 years later?. Proceedings of the 16th Arctic and Marine Oilspill Programme (AMOP) Technical Seminar. Environment Canada, Ottawa, On. p1149-1167.

Owens, E.H., Prince, R.C. and R.B. Taylor. 2008. Natural attenuation of heavy oil on a coarse sediment beach: results from Black Duck Cove, Nova Scotia, Canada over 35 years following the *Arrow* oil spill. In: *Proceedings of the 31st AMOP technical seminar on environmental contamination and response.*

Owens, E.H. 2010. Shoreline response and long-term oil behaviour studies following the 1970 "*Arrow*" spill in Chedabucto Bay, NS. Proceedings of the 33rd Arctic and Marine Oilspill Programme (AMOP) Technical Seminar. Environment Canada, Ottawa, On.

Paris, C. B., Le Hénaff, M., Aman, Z.M., Subramaniam, A., Helgers, J., Wang, D. -P.,Kourafalou, V.H. and A. Srinivasan. 2012. Evolution of the Macondo Well blowout: Simulating the effects of the circulation and synthetic dispersants on the subsea oil transport. *Environmental Science & Technology* 46(24): 13293-13302.

Parker-Hall, H.A. and E.H. Owens. 2006. Lake Wabamun Derailment: Fate and Persistence of the Spilled Oil. Polaris Applied Sciences. Prepared for CN. 38 pp.

Peterson, C.H., Rice, S.D., Short,J.W. Esler, D., Bodkin, J.L., Ballachey, B.E. and D.B. Irons. 2003. Long-term ecosystem response to the *Exxon Valdez* oil spill. *Science* 302: 2082-2086.

Peterson, C. H., Anderson, S.S., Cherr, G.N., Ambrose, R.F., Anghera, S. Bay, S., Blum, M., Condon, R., Dean, T.A., Graham, M. Guzy, M., Hampton, S. Joye, S., Lambrinos, J., Mate, B., Meffert, D., Powers, S.P., Somasundaran, P., Spies, R.B., Taylor, C.M., Tjeerdema, R. and E.E. Adams. 2012. A tale of two spills: novel science and policy implications of an emerging new oil spill model. *Bioscience* 62: 461-469.

Platt, J.F. and R. G. Ford. 1996. How many seabirds were killed by the *Exxon Valdez* spill? *American Fisheries Society Symposium* 18: 712-719.

Prepas, E. and P. Mitchell. 1990. Atlas of Alberta Lakes. University of Alberta Press. Digital edition available at: http://sunsite.ualberta.ca/Projects/Alberta-Lakes/preface.php

Prince, R.C. 2015. Oil spill dispersants; bone or bane? *Environmental Science & Technology* 49: 6376-6384.

Prince, R.C., Garrett, R. M., Bare, R.E., Grossman, M.J., Townsend, T., Suflita, J.M., Lee, K., Owens, E.H., Sergy, G.A., Braddock, J.F., Lindstrom, J.E. and R. R. Lessard. 2003. The roles of photooxidation and biodegradation in long-term weathering of crude and heavy fuel oils. *Spill Science and Technology Bulletin* 8: 145-156.

Pritchard, P.H., Mueller, J.G., Rogers, J.C., Kremer, F.V. and J.A. Glaser. 1992. Oil spill bioremediation: experiences, lessons and results from the *Exxon Valdex* oil spill in Alaska. *Biodegradation* 3: 315-335.

Reddy, C. M., Arey, J. S., Seewald, J.S., Sylva, S.P., Lemkau, K.L., Nelson, R.K., Carmichael, C.A., McIntyre, C.P., Fenwick, J. and G. T. Ventura. 2011. Composition and fate of gas and oil released to the water column during the Deepwater Horizon oil spill. *Proceedings of the National Academy of Sciences of the United States of America* 109:20229-20234.

ED_001324_00007647-00456

Redmond, M.C. and D.L. Valentine. 2012. Natural gas and temperature structured a microbial community response to the *Deepwater Horizon* oil spill. *Proceedings of the National Academy of Sciences* 109: 20292-20297.

Reich, D.A., Balouskus, R., French-McCay, D., Fontenault, J. Rowe, J., Singer-Leavitt, Z., Schmidt Erkin, D., Michel, J., Nixon, Z., Boring, C., McBrien, M. and B. Hay. 2014. Assessment of Marine Oil Spill Risk and Environmental Vulnerability for the State of Alaska. Submitted to the National Oceanic and Atmospheric Administration, Seattle, WA. NOAA Contract Number: WC133F-11-CQ-0002. https://alaskafisheries.noaa.gov/habitat/restoration/oilspill/riskreport.pdf

Rico-Martinez, R., Snell, T.W., and T.L. Shearer. 2013. Synergistic toxicity of Macondo crude oil and dispersant Corexit 9500A® to the *Brachionus plicatilis* species complex (Rotifera). *Environmental Pollution* 173: 5-10.

Schindler, D.W., Anderson, A.-M., Brzustowski, J., Donahue, W.F., Goss, G., Nelson, J., St. Louis, V., Sullivan, M. and S. Swanson. 2004. Lake Wabamun:A Review of Scientific Studies and Environmental Impacts. Submitted to the Minister of Alberta Environment, Edmonton, AB. http://www3.gov.ab.ca/env/water/reports/wabamun/Wabamun_Report_Dec04.pdf.

Sempels, J.-M. 1982. Coastlines of the Eastern Arctic. Arctic 35: 170-179.

Sergy, G.A and P.J. Blackall. 1987. Design and conclusions of the Baffin Island Oil Spill Project. Arctic 40 (Supp 1): 1-9.

Shigenaka, G. 2014. Twenty-Five Years After the *Exxon Valdez* Oil Spill: NOAA's Scientific Support, Monitoring, and Research. Seattle: NOAA Office of Response and Restoration. http://response.restoration.noaa.gov/sites/default/files/Exxon_Valdez_25YearsAfter_508_0.pdf

Short, J. W., M. R. Lindeberg, P. A. Harris, et al. 2004. Estimate of oil persisting on beaches of Prince William Sound, 12 years after the *Exxon Valdez* oil spill. *Environmental Science & Technology* 38:19–25.

Short, J.W. 2008. An Evaluation of Oil Fate, Persistence and Toxic Potential to Fish Following the August 2005 Canadian National Derailment Spill into Lake Wabamun, Alberta. August 27, 2008. https://docs.neb-one.gc.ca/ll-eng/llisapi.dll/Open/2586376.

Short, J.W. 2015. Presentation to the Royal Society Expert Panel on the Behaviour and Environmental Impacts of Crude Oil Released in Aqueous Environments. April, 2015, Halifax, NS.

Silliman, B.R., van de Koppel, J., McCoy, M.W., Diller, J., Kasozi, G.N., Earl, K., Adamns, P.N. and A.R. Zimmerman. 2012. Degradation and resilience in Louisiana salt marshes after the BP-*Deepwater Horizon* oil spill. *Proceedings of the National Academy of Sciences* 109: 11234-11239.

Spies, R.B., Rice, S.D., Wolfe, D.A. and B.A. Wright. 1996. The effect of the Exxon Valdez oil spill on Alaskan coastal environment. In: *Proceedings of the 1993 Exxon Valdez Oil Spill Symposium*. American Fisheries Society, Bethesda, MD. Pp. 1-16

SL Ross. 2014. Guideline Document for Measuring the Biological Effects of Accidental Oil Spills. Submitted to Fisheries and Oceans Canada, National Contaminants Advisory Group, Ottawa, ON.

Stanbury, M., Hekman, K., Wells, E., Miller, C., Smolinske, S. and J. Rutherford. 2010. Acute Health Effects of the Enbridge Oil Spill. Michigan Department of Community Health, Lansing, MI. http://www.michigan.gov/documents/mdch/enbridge_oil_spill_epi_report_with_cover_11_22_10_339101_7.pdf.

Stantec. 2012. Summary of Clean Up and Effects of the 2007 Spill of Oil from Trans Mountain Pipeline to Burrard Inlet. http://www.transmountain.com/uploads/pages/1374960772-2012-Summary-2007-Spill-Cleanup---Effects-REV2.pdf

State of Alaska. 1993. Shoreline treatment techniques. In: *The Exxon Valdez Oil Spill: Final Report, State of Alaska Response*. Alaska Department of Environmental Conservation. Pp. 61-87.

Stephenson, M., Henderson, J., Mazzacco, P. and A. St-Amand. 2015. Ecological risk assessment of hypothetical spills of diluted bitumen in rivers. In: *Proceedings of the Thirty-Eight AMOP Technical Seminar.* Environment Canada, Ottawa, ON, pp. 607-630.

Sumners, J. 2001. July 31 2000 Pine River Oil Spill Physical Fish Habitat Impact Statement. Submitted to John Cliffe, Crown Counsel, Department of Justice.

Tarnecki, J.H. and W.F. Patterson III. 2015. Changes in red snapper diet and trophic ecology following the Deepwater Horizon oil spill. *Marine and Coastal Fisheries* 7: 135-147

Thomas, M.L. 1978. Comparison of oiled and unoiled intertidal communities in Chedabucto Bay, Nova Scotia. *Journal of the Fisheries Research Board of Canada* 35(5): 707-716.

Thormann, M.N. and S.E. Bayley. 2008. Impacts of the CN Rail Oil Spill on Softstem Bulrush-Dominated Lacustrine Marshes in Wabamun Lake. Report submitted to Alberta Environment, Edmonton, AB.

TSB (Transportation Safety Board of Canada). 2007. Pipeline Investigation Report: Crude oil Pipeline – Third-Party Damage. Trans Mountain Pipeline L.P. 610-millimetre-diameter crude oil pipeline. Kilometre post 3.10, Westridge Dock Transfer Line, Burnaby, BC. Report P07H0040.

USGS. 2015. Research, Response for Future Oil Spills: Lessons Learned from Deepwater Horizon. http://www.usgs.gov/newsroom/article.asp?ID=3469&from=rss#.VeUu9ZWFOpo

Valentine, D. L. Fisher, G.B., Bagby, S.C., Nelson, R.K., Reddy, C.M., Sylva, S.P. and M.A. Woo. 2014. Fallout plume of submerged oil from *Deepwater Horizon*. P*roceedings of the Nationa. Academy of Sciences USA*. http://www.pnas.org/cgi/doi/10.1073/pnas.1414873111

Vancouver Sun. 2013. CN's crude oil-by-rail plan increases risk of spills, fatalities, say opponents. Vancouver Sun February 1, 2013. http://www.canada.com/story.html?id=bed39d62-7e95-498b-be36-b8f8e0913089

Vandermeulen, J.H. and J.G. Singh. 1994. *Arrow* oil spill, 1970-90: persistence of 20-yr weathered Bunker C fuel oil. *Canadian Journal of Fisheries and Aquatic Sciences* 51: 845-855.

Venn-Watson S, Colegrove KM, Litz J, Kinsel M, Terio K, Saliki J, et al. (2015) Adrenal Gland and Lung Lesions in Gulf of Mexico Common Bottlenose Dolphins (*Tursiops truncatus*) Found Dead following the *Deepwater Horizon* Oil Spill. PLoS ONE 10(5): e0126538. doi:10.1371/journal.pone.0126538

Wang, Z., M.F. Fingas and G. Sergy. 1994. Study of 22-Year-Old ARROW Oil Samples Using Biomarker Compounds by GC/MS. *Environmental Science and Technology*, 28:1733-1746.

Water Survey of Canada. 2015. Discharge Graph for Pine River at East Pine (07FB001). https://wateroffice.ec.gc.ca/report/report_e.html?mode=Graph&type=h2oArc&stn=07FB001&dataType=Daily&parameterType=Flow&year=2012&y1Max=1&y1Min=1&y1Mean=1&scale=normal

Weinberg, A. 1972. Science and trans-science. Science 21:209-22.

Wernick, B., deBruyn, A.H., Patterson, L. and P.Chapman. 2009. Effects of an oil spill on the regrowth of emergent vegetation in a Northern Alberta lake. *Archives of Environmental Contamination and Toxicology* 57: 697-706.

White, H.K, Hsing, P-Y., Cho, W., Shank, T.N., Cordes, E.E., Quattrin, A.M., Nelson, R.K., Camilli, R., Demopoiulos, A.W.J., German, C.E., Brooks, J.M., Roberts, H.H., Shedd, W., Reddy, C.M. and C.R. Fisher. 2012. Impacts of the Deepwater Horizon oil spill on a deep-water coral community in the Gulf of Mexico. *Proceedings of the National Academy of Sciences* 109: 20303-20308.

Wolfe DA, Hameedi MJ, Galt JA, Watabaytashi G, Short J, O'Clair C, Rice S, Michel J, Payne JR, Braddock J, Hanna S, Sale D. 1994. The fate of the oil spilled from the Exxon Valdez. Environ Sci Technol 28:561–568.

WSP and SL Ross. 2014a. Risk Assessment for Marine Spill in Canadian Waters. Phase 1: Oil Spills South of 60th Parallel. Final Study Report prepared for Transport Canada. http://wcel.org/sites/default/files/file-downloads/131-17593-00_ERA_Oil-Spill-South_150116_pp1-124.pdf.

WSP and SL Ross. 2014b. Risk Assessment for Marine Spills in Canadian Waters. Phase 2, Part B: Spills of Oil and Select Hazardous and Noxious Substances (HNS) Transported in Bulk North of the 60th Parallel North. Final Report prepared for Transport Canada.

Appendix A

ASTM (American Society of Testing and Materials). 2011. Method D7678 – 11. Standard Test Method for Total Petroleum Hydrocarbons (TPH) in Water and Wastewater with Solvent Extraction Using Mid-IR Laser Spectroscopy. DOI: 10.1520/D7678-11. http://www.astm.org/Standards/D7678.htm.

Brown, C. W., Lynch, P. F. and M. Ahmadjian. 1975. Applications of infrared spectroscopy in petroleum analysis and oil spill identification. *Applied Spectroscopy Reviews* 9: 223-248.

Butt, J.A., Duckworth, D. and S.G. Perry. 1986. Characterization of Spilled Oil Samples: Purpose, Sampling, Analysis and Interpretation. Institute of Petroleum. John Wiley & Sons, London.

de Andrade, D. F., Fernandes, D. R. and J.L. Miranda. 2010. Methods for the determination of conjugated dienes in petroleum products: A review. *Fuel* 89: 1796-1805.

Eide, I. and K. Zahlsen. 2005. A novel method for chemical fingerprinting of oil and petroleum products based on electrospray mass spectrometry and chemometrics. *Energy and Fuels* 19: 964-967.

Fernández-Varela, R., Gómez-Carracedo, M. P., Fresco-Rivera, P., Andrade, J. M., Muniategui, S. and D. Prada. 2006. Monitoring photooxidation of the *Prestige's* oil spill by attenuated total reflectance infrared spectroscopy. *Talanta* 69: 409-417.

Fish, R. H., Komlenic, J. J. and B.K. Wines. 1984. Characterization and comparison of vanadyl and nickel compounds in heavy crude petroleums and asphaltenes by reverse-phase and size-exclusion liquid chromatography/graphite furnace atomic absorption spectrometry. *Analytical Chemistry* 56: 2452-2460.

Gaines, R. B., Frysinger, G. S.; Hendrick-Smith, M. S. and K.D. Stuart. 1999. Oil spill source identification by comprehensive two-dimensional gas chromatography. *Environmental Science & Technology* 33: 2106-2112.

Gautam, K., Jin, X. and M. Hansen. 1998. Review of spectrometric techniques for the characterization of crude oil and petroleum products. *Applied Spectroscopy Reviews* 33: 427-443.

Kenkel, J. 2002. Analytical Chemistry for Technicians. CRC Press, London.

Khanmohammadi, M., Garmarudi, A.B. and M. de la Guardia. 2012. Characterization of petroleum-based products by infrared spectroscopy and chemometrics. *TrAC Trends in Analytical Chemistry* 35: 135–149

Kim, M., Hong, S. H., Won, J., Yim, U. H., Jung, J. H., Ha, S. Y., An, J. G., Joo, C., Kim, E., Han, G. M., Baek, S., Choi, H. W. and W. J. Shim. 2013. Petroleum hydrocarbon contaminations in the intertidal seawater after the *Hebei Spirit* oil spill – Effect of tidal cycle on the TPH concentrations and the chromatographic characterization of seawater extracts. *Water Research* 47: 758-768.

Li, K. and M. Fingas. 1997. An Evaluation of Bench Top Liquid Chromatography/Solid Probe Mass Spectrometry for Emergency Spill Sample Analysis. Environment Canada, Ottawa.

Lynch, P. F. and C.W. Brown. 1973. Identifying source of petroleum by infrared spectroscopy. *Environmental Science & Technology* 7: 1123-1127.

Pasadakis, N., Gaganis, V. and N. Varotsis. 2001. Accurate determination of aromatic groups in heavy petroleum fractions using HPLC-UV-DAD. *Fuel* 80: 147-153.

Payne, J.R., Driskell, W.B., Short, J.W. and M.L. Larsen. 2008. Long term monitoring for oil in the *Exxon Valdez* spill region. *Marine Pollution Bulletin* 56: 2067-2081.

Rogers, R.P. and A.M. McKenna. 2011. Petroleum analysis. *Analytical Chemistry.* 83: 4665-4687.

Rostad, C. E. 2006. Differentiation of commercial fuels based on polar components using negative electrospray ionization/mass spectrometry. *Environmental Forensics* 7: 5-14.

Roussis, S.G. and W.P. Fitzgerald. 2000. Gas chromatographic simulated distillation-mass spectrometry for the determination of the boiling point distributions of crude oils. *Analytical Chemistry* 72: 1400-1409.

Simanzhenkov, V. and R. Idem. 2003. Crude Oil Chemistry. Marcel Dekker, New York.

Speight, J. G. and J. Speight. 2002. Handbook of Petroleum Product Analysis. Wiley-Interscience, New Jersey.

Steffens, J., Landulfo, E., Courrol, L.C. and R. Guardani. 2011. Application of fluorescence to the study of crude petroleum. *Journal of Fluorescence* 21: 859-864.

Sun, P. Y., Bao, M. T., Li, G. M., Wang, X. P., Zhao, Y. H., Zhou, Q. and L.X. Cao. 2009. Fingerprinting and source identification of an oil spill in China Bohai Sea by gas chromatography-flame ionization detection and gas chromatography-mass spectrometry coupled with multi-statistical analyses. *Journal of Chromatography A* 1216: 830-836.

Tzing, S. H., Chang, J. Y., Ghule, A., Chang, J. J., Lo, B. and Y.C. Ling. 2003. A simple and rapid method for identifying the source of spilled oil using an electronic nose: confirmation by gas chromatography with mass spectrometry. *Rapid Communications in Mass Spectrometry* 17: 1873-1880.

Wang, Z., Li, K., Fingas, M., Sigouin, L. and L. Ménard. 2002. Characterization and source identification of hydrocarbons in water samples using multiple analytical techniques. *Journal of Chromatography A* 971: 173-184.

Yim, U. H., Ha, S. Y., An, J. G., Won, J. H., Han, G. M., Hong, S. H., Kim, M., Jung, J. H. and W.J. Shim. 2011. Fingerprint and weathering characteristics of stranded oils after the *Hebei Spirit* oil spill. *Journal of Hazardous Materials* 197: 60-69.

Appendix B

Aske, N., Kallevik, H. and J. Sjöblom. 2002. Water-in-crude oil emulsion stability studied by critical electric field measurements. Correlation to physico-chemical parameters and near-infrared spectroscopy. *Journal of Petroleum Science and Engineering* 36: 1-17.

ASTM (The American Society for Testing and Materials). 2005. ASTM D5002-99: Standard Test Method for Density and Relative Density of Crude Oils by Digital Density Analyzer. DOI: 10.1520/D5002-99R05.

ASTM. 2007. ASTM D1310 – 01: Standard Test Method for Flash Point and Fire Point of Liquids by Tag Open-Cup Apparatus. DOI: 10.1520/D1310-01R07.

ASTM. 2012a. ASTM D97 – 12: Standard Test Method for Pour Point of Petroleum Products. DOI: 10.1520/D0097-12.

ASTM. 2012b. ASTM D287 – 12: Standard Test Method for API Gravity of Crude Petroleum and Petroleum Products (Hydrometer Method). DOI: 10.1520/D0287-12B.

ASTM. 2013. ASTM D7566 - 13 Standard Specification for Aviation Turbine Fuel Containing Synthesized Hydrocarbons. DOI: 10.1520/D7566-13

ASTM. 2015. ASTM D445-15 Standard Test Method for Kinematic Viscosity of Transparent and Opaque Liquids (and Calculation of Dynamic Viscosity). DOI: 10.1520/D0445-15

Beal, C. 1946. The viscosity of air, water, natural gas, crude oil and its associated gases at oil field temperatures and pressures. *Transactions of the AIME* 165: 94-115.

Beggs, H. D. and J. Robinson, J. 1975. Estimating the viscosity of crude oil systems. *Journal of Petroleum Technology* 27: 1140-1141.

du Noüy, P. L. 1925. An interfacial tensiometer for universal use. *The Journal of General Physiology* 7: 625-631.

Fingas, M., Hollebone, B. and B. Fieldhouse. 2006. The density behaviour of heavy oils in freshwater: The example of the Lake Wabamun spill. In: *Proceedings of the 29th Arctic and Marine Oilspill Program.* Environment Canada, Ottawa, ON. pp. 57-77.

Fingas, M. F. 2010. Review of the North Slope Oil Properties Relevant to Environmental Assessment and Prediction. Report prepared for Prince William Sound Regional Citizens' Advisory Council (PWSRCAC). Anchorage, Alaska.

Lambert, P. 2003. A literature review of portable fluorescence-based oil-in-water monitors. *Journal of Hazardous Materials* 102(1): 39-55.

Morrison, R. D. and B.L. Murphy. 2006. Environmental Forensics: Contaminant Specific Guide. Academic Press.

Appendix C

American Fuel & Petrochemical Manufacturers. 2014. A Survey of Bakken Crude Oil Characteristics Assembled for the U.S. Department of Transportation. http://www.ourenergypolicy.org/wp-

ED_001324_00007647-00460

content/uploads/2015/06/Survey-of-Crude-Oil-Characteristics_FINAL-1.pdf (accessed June, 2015)

BP (British Petroleum) Gulf Science Data MC-252 Oil Characterization Data File. Reference No. O-01v01-01. Last modified November 12, 2013. http://gulfsciencedata.bp.com/go/doctype/6145/178706 (accessed June, 2015)

Crude Quality Inc. 2015. crudemonitor.ca. http://crudemonitor.ca. (accessed June 2015)

Enbridge Pipelines Inc. 2014 Crude Characteristics No. 45. Enbridge Energy Partners, L.P. *http://www.enbridge.com/~/media/www/Site%20Documents/Delivering%20Energy/2014%20Mainline%20Crude%20Characteristics.pdf?la=en*

Environmental Technology Centre, Environment Canada. 2015. Oil Properties Database. http://www.etc-cte.ec.gc.ca/databases/oilproperties/. (accessed June 2015)

Hollebone, B.P. 2015. Appendix A. The oil properties data appendix. In: M.F. Fingas, M.F.(ed.) *Handbook of Oil Spill Science and Technology*. Wiley & Sons, Inc., Hoboken, NJ.

Indian and Northern Affairs Canada. 1995. Petroleum Exploration in Canada. Chapter 4 – Mackenzie Delta and Beauford Sea. http://www.aadnc-aandc.gc.ca/eng/1321378288752/1321378450630 (accessed June 2015)

MSDS product data sheets https://www.msdsonline.com/msds-search

North Dakota Petroleum Council. No date. Bakken Crude Properies. http://www.ndoil.org/resources/bkn/. (accessed June 2015)

National Research Council. 2013. TRB Special Report 311: Effects of Diluted Bitumen on Crude Oil Transmission Pipelines. http://www.nap.edu/read/18381/chapter/1 (accessed Oct 2015)

Wikipedia. 2015. Western Canadian Select. http://en.wikipedia.org/wiki/Western_Canadian_Select. (accessed June 2015)

To: Vanessa Principe[veprincipe5@verizon.net]
From: Principe, Vanessa
Sent: Wed 11/25/2015 7:31:50 PM
Subject: FW: State-of-Science: Public Health and Food Security follow-up and REPLY
2013 D'Andrea paper.pdf
Health-Risks-Associated-with-Crude-Oil-Spill-Exposure_2014_The-American-Journal-of-Medicine.pdf
VirusWarning.txt
The Reply to Pacentino.pdf
The-Reply_2014_Wickliffe_The-American-Journal-of-Medicine.pdf
Wickcliffe response.pdf
2015.11.23_Public Health and Food Security.docx
Suggestions from John French_new concerns and references.docx

From: Mandsager, Kathy [mailto:kathy.mandsager@unh.edu]
Sent: Wednesday, November 25, 2015 2:29 PM
To: jberner@anmc.org; sandrine.deglin@alaska.gov; robert.dickey@utexas.edu; jim.fall@alaska.gov; PESA@gci.net; jgohlke@vt.edu; sklasing@oehha.ca.gov; richard.kwok@nih.gov; ken.lindemann@bp.com
Cc: nancy.kinner@unh.edu; Gary Shigenaka - NOAA Federal <gary.shigenaka@noaa.gov>; doug.helton@noaa.gov; Principe, Vanessa <Principe.Vanessa@epa.gov>; Wilson, Gregory <Wilson.Gregory@epa.gov>
Subject: State-of-Science: Public Health and Food Security follow-up and REPLY

As follow-up to our call on 11/23, please find information below and attached:

•□□□□□□□□ Here is the link for the Federal Registry for "Guidelines for Ensuring and Maximizing the Quality, Objectivity, Utility, and Integrity of Information Disseminated" https://www.whitehouse.gov/sites/default/files/omb/fedreg/reproducible2.pdf

•□□□□□□□□ Attached are: D'Andrea paper 2013 ("Health Consequences among Subjects Involved in Gulf Oil Spill Clean-up Activities") and 2014 ("Health Risks Associated with Crude Oil Spill Exposure"); Two responses from Piacentino and Wickliffe; and the author's reply to these 2 comments.

•□□□□□□□□ I am also attaching once again the latest draft of our document, along with the proposed wording & resources from John French.

And here is a doodle poll to select our next meeting>> http://doodle.com/poll/59ctafsri9zzgvzm
Please fill out by Monday 11/30 for these potential December meeting dates. The doodle poll is time zone enabled for your convenience.

Kathy Mandsager

Program Coordinator

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THE AMERICAN JOURNAL of MEDICINE

Health Consequences among Subjects Involved in Gulf Oil Spill Clean-up Activities

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University Cancer and Diagnostic Centers, Houston, Tex.

ABSTRACT

BACKGROUND: Oil spills are known to affect human health through the exposure of inherent hazardous chemicals such as para-phenols and volatile benzene. This study assessed the adverse health effects of the Gulf oil spill exposure in subjects participating in the clean-up activity along the coast of Louisiana.
METHODS: This retrospective study included subjects that had been exposed and unexposed to the oil spill and dispersant. Using medical charts, clinical data including white blood cell count, platelets count, hemoglobin, hematocrit, blood urea nitrogen, creatinine, alkaline phosphatase (ALP), aspartate amino transferase (AST), alanine amino transferase (ALT), and somatic symptom complaints by the subjects were reviewed and analyzed.
RESULTS: A total of 247 subjects (oil spill exposed, n ¼ 117 and unexposed, n ¼ 130) were included. Hematologic analysis showed that platelet counts (┐ 10^3 per mL) were significantly decreased in the exposed group compared with those in the group unexposed to the oil spill (252.1 └ 51.8 vs 269.6 └ 77.3, P ¼ .024). Conversely, the hemoglobin (g per dL) and hematocrit (%) levels were significantly increased among oil spill-exposed subjects compared with the unexposed subjects (P ¼ .000). Similarly, oil spill-exposed subjects had significantly higher levels of ALP (76.3 └ 21.3 vs 61.2 └ 26.9 IU/L, P ¼ .000), AST (31.0 └ 26.3 vs 22.8 └ 11.8 IU/L, P ¼ .004), and ALT (34.8 └ 26.6 vs 29.8 └ 27 IU/L, P ¼ .054) compared with the unexposed subjects.
CONCLUSION: The results of this study indicate that clean-up workers exposed to the oil spill and dispersant experienced significantly altered blood profiles, liver enzymes, and somatic symptoms.
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KEYWORDS: Blood disorders; Chemical exposure; Crude oil spill; Dispersants; Health impact; Hematological toxicity; Hepatotoxicity; Somatic symptoms; Urinary phenol

Crude oil spills affect human health through exposure to inherent hazardous chemicals including para-phenols and volatile benzene.[1-3] The major health consequences of crude oil spill exposure include the abnormalities in hematologic, hepatic, respiratory, renal, and neurologic functions.[4,5] In addition, subjects exposed to oil spills often experience frequent asthmatic attacks, headache, diarrhea, dizziness, abdominal pain, back pain, and other symptoms.[4,6-9]

On April 20, 2010, the British Petroleum (BP) Deepwater Horizon offshore drilling rig located 50 miles off the Louisiana coast exploded and sank in the Gulf of Mexico (Figure 1).[5,10] Consequently over 200 million gallons of oil poured into the Gulf of Mexico,[11] thereby contaminating the Gulf coast. During the height of this disaster, BP used nearly 2 million gallons of dispersants such as COREXIT[12] (Nalco Energy Services, L.P., Sugar Land, Tex) to break down the oil slick.[11] This oil spill and use of massive amounts of dispersant has the potential to affect human health. It is estimated that up to 170,000 people worked in some capacity to clean up the Gulf oil spill.[13]

Previously, several studies have evaluated the health impact of other oil spills.[5,6,10,14-16] These studies primarily focused on physical effects and psychological sequelae. In addition, these studies point to potential adverse effects among oil spill clean-up workers. Earlier studies reported that benzene exposure is associated with hematological toxicity and increased cancer risk.[17-22] While such rare adverse outcomes may take years to develop, immediate health effects of

Funding: None.
Conflict of Interest: None.
Authorship: Both authors listed in the manuscript had access to the data and a role in preparing the manuscript.
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http://dx.doi.org/10.1016/j.amjmed.2013.05.014

oil spill exposure may be seen in hematological and hepatic parameters, indicating its toxic effects and potential for future health risk.[23]

To investigate the adverse health effects of the oil spill in the Gulf of Mexico, a retrospective analysis was performed on subjects participating in the oil spill clean-up activity along the coast of Louisiana. Specifically, we assessed the hematologic and hepatic markers in a cohort of oil spill clean-up workers, and the clinical findings were compared with the unexposed (control) subjects.

CLINICAL SIGNIFICANCE

- Human exposure to oil spill and dispersant use has a potential to alter both hematological and hepatic profiles.
- The hematological and hepatic alterations include decreased platelets and BUN, and increased creatinine, serum levels of ALP, AST, and ALT.
- The most reported somatic symptoms are headache, shortness of breath, skin rash, cough, fatigue, painful joints, and chest pain.

MATERIALS AND METHODS

Subjects

This study was approved by an Institutional Review Board. Using medical charts, demographic and clinical data were reviewed for the subjects who underwent clinical and laboratory evaluation between January 2010 and November 2012. The study was conducted according to the principles of the Declaration of Helsinki. The personal information of the subjects was redacted to maintain confidentiality.

Subjects included in this study were referred to the clinic for medical evaluation by the subjects' legal representatives. Written consent was obtained from the subjects' legal representatives. The subjects exposed to the oil spill and dispersant were identified as participants in the oil spill clean-up activities along the coast of Louisiana for a duration of over 3 months.

The unexposed subjects were living geographically at least 100 miles away from the Gulf coast of Louisiana. The unexposed subjects had visited the clinic for a routine wellness check-up. The subjects were selected randomly for the study by their primary care physicians.

Chart Review and Evaluation

Medical charts of exposed and unexposed subjects were reviewed by expert physicians and the data were processed for statistical analysis. Clinical data such as white blood cell (WBC) counts, platelet counts, hemoglobin, hematocrit, blood urea nitrogen (BUN), creatinine, serum beta-2-microglobulin, alkaline phosphatase (ALP), aspartate amino transferase (AST), and alanine amino transferase (ALT) levels were evaluated. Data on urinary phenol also was assessed as a benzene metabolite in oil spill-exposed subjects. Additionally, data on somatic symptoms were collected from the oil spill-exposed subjects and analyzed.

Statistics

Descriptive statistics were used to assess patient demographics and included means and standard deviations for each group. Variables included were WBC, platelets, hemoglobin, hematocrit, creatinine, BUN, ALP, AST, ALT, beta-2 microglobulin, and urinary phenol. Student's t test was used to assess the differences between exposed and unexposed groups. The significance level was predetermined at an alpha level of .05.

RESULTS

This study included a total of 247 subjects, 117 of which were involved in the clean-up activity of the oil spill. The outcomes of the oil spill-exposed subjects were evaluated and compared with the unexposed subjects (n ¼ 130). The subjects' demographics are shown in Table 1. Among the oil spill-exposed subjects (n ¼ 117), there were 104 (89%) males and 13 (11%) females. The median age of exposed and unexposed subjects was 34.0 (18-63) and 51.0 (15-90) years, respectively. The demographic characteristics such as sex and age groups differed significantly between the exposed and unexposed groups (P ¼ .000).

The data in Table 2 represents the differences in hematologic and hepatic markers between the subjects exposed to the oil spill and those unexposed. No significant differences were observed in the WBC count (¬ 10^3 per mL) among those exposed to the oil spill and those unexposed (6.9 └ 1.9 vs 6.5 └ 1.9, P ¼ .11). However, the platelet count (¬ 10^3 per mL) in subjects exposed to the oil spill was significantly decreased compared with the unexposed subjects (252.1 └ 51.8 vs 269.6 └ 77.3, P ¼ .024).

The hemoglobin (g per dL) levels were significantly increased among oil spill-exposed subjects compared with the unexposed subjects (14.9 └ 1.3 vs 13.6 └ 1.6, P ¼ .000). The hematocrit levels were significantly elevated in oil spill-exposed subjects compared with the unexposed subjects (44.6 └ 3.4 vs 40.8 └ 5.0, P ¼ .000). Similarly, serum creatinine levels (mg/dL) also were significantly higher in the oil spill-exposed subjects compared with the unexposed subjects (1.0 └ 0.2 vs 0.9 └ 0.3, P ¼ .000). Conversely, the BUN (mg/dL) significantly decreased in the oil spill-exposed subjects compared with the unexposed subjects (13.4 └ 3.4 vs 15.4 └ 7.1, P ¼ .014).

Compared with the unexposed subjects, subjects exposed to the oil spill showed significantly elevated levels of ALP (76.3 └ 21.3 vs 61.2 └ 26.9 IU/L, P ¼ .000). The AST (IU/L) levels were significantly higher in the oil spill-exposed subjects compared with the unexposed subjects (31.0 └ 26.3 vs 22.8 └ 11.8, P ¼ .004). Similarly, the ALT (IU/L) levels were marginally but significantly increased in the oil spill-exposed group compared with the unexposed group (34.8 └ 26.6 vs 29.8 └ 27, P ¼ .054).

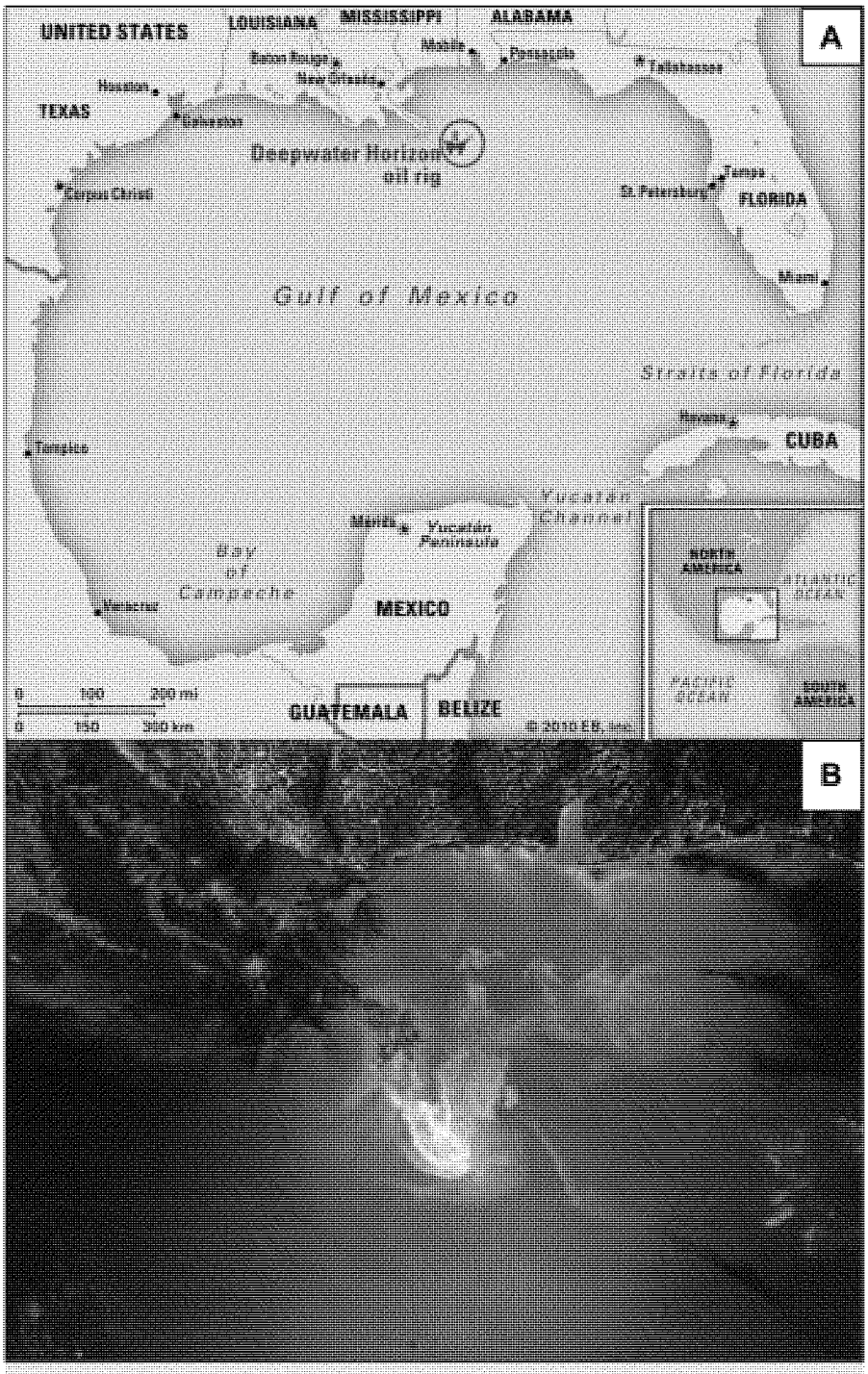

Figure 1 The Deepwater Horizon oil drilling rig. (A) The location of the Deepwater Horizon oil rig explosion in the Gulf of Mexico, 50 miles off the Louisiana coast (Source: http://www.britannica.com/blogs/2011/04/deepwater-horizon-oil-spill-year-picture-essay-day/ By courtesy of Encyclopedia Britannica, Inc., Copyright 2010; used with permission). (B) Satellite view of the Deepwater Horizon oil spill slick by the US National Aeronautics and Space Administration in the Gulf of Mexico (Source: http://www.nasa.gov/topics/earth/features/oilspill/index.html. Accessed January 15, 2013).

The results in Table 3 show the differences in hematologic and hepatic markers between exposed and unexposed subjects according to their age group. Specifically, subjects were grouped into <40-year and ≥ 40-year age groups, and clinical outcomes were compared. No differences were noted in the WBC count (≥ 10^3 per mL) between unexposed and exposed subjects in the <40-year age group; however, the WBC counts (≥ 10^3 per mL) were significantly higher in

Table 1 Demographics and Other Characteristics of the Subjects Included in the Study

Demographics	Unexposed	Exposed	P Value
Total subjects	130 (100%)	117 (100%)	-
Sex			.000*
Male	43 (33%)	104 (89%)	
Female	87 (67%)	13 (11%)	
Age, years			.000*
Mean	49.8	35.8	
Median	51	34	
Range	15-90	18-63	
Age group, years			.000*
<20	4 (3%)	0 (0%)	
20-<30	13 (10%)	31 (27%)	
30-<40	12 (9%)	42 (36%)	
40-<50	28 (22%)	35 (30%)	
50-<60	41 (32%)	8 (7%)	
60-<70	20 (15%)	1 (1%)	
70	12 (9%)	0 (0%)	

*Statistically significant at P ¼ .001.

oil spill-exposed subjects compared with the unexposed subjects in the 40-year age group (7.5 2.0 vs 6.5 1.9, P ¼ .007). Similarly, the platelet counts (10^3 per mL) were significantly higher in oil spill-exposed subjects compared with the unexposed subjects in the – 40-year age group (283.6 67.3 vs 251.8 51.7, P ¼ .006). However, no differences were noted in the platelet counts between exposed and unexposed subjects in the <40-year age group. Hemoglobin (g per dL) and creatinine (mg per dL) were increased significantly in oil spill-exposed subjects compared with the unexposed subjects, irrespective of age groups. Similarly, the serum levels of hepatic enzymes (ALP, AST, and ALT) were increased significantly in oil spill-exposed subjects compared with those unexposed subjects, irrespective of age groups (Table 3).

Table 2 Comparison of Hematologic and Hepatic Indices between the Unexposed and Exposed Subjects to Oil Spill

Variable	Unexposed (n ¼ 130)	Exposed (n ¼ 117)	P Value
WBC (10^3 per mL)	6.5 1.9	6.9 1.9	.108†
Platelets (10^3 per mL)	269.6 77.3	252.1 51.8	.024*
Hemoglobin (g per dL)	13.6 1.6	14.9 1.3	.000**
Hematocrit (%)	40.8 5.0	44.6 3.4	.000**
BUN (mg per dL)	15.4 7.1	13.4 3.4	.014*
Creatinine (mg per dL)	0.9 0.3	1.0 0.2	.000**
ALP (IU per L)	61.2 26.9	76.3 21.3	.000**
AST (IU per L)	22.8 15.1	31.0 26.3	.004**
ALT (IU per L)	29.8 27.0	34.8 26.6	.054*

ALP ¼ alkaline phosphatase; ALT ¼ alanine amino transferase; AST ¼ aspartate amino transferase; BUN ¼ blood urea nitrogen; WBC ¼ white blood cells.

*P ¼ 0.05.

**P ¼ 0.001.

†Did not reach statistical significance.

The results in Table 4 indicate the differences in hematologic and hepatic markers between the unexposed subjects and those exposed to the oil spill according to sex. No differences were noted in WBC counts (10^3 per mL) between unexposed and exposed male subjects; however, female subjects in the oil spill-exposed group had significantly elevated WBC counts (10^3 per mL) compared with the unexposed group (7.5 2.2 vs 6.4 1.7, P ¼ .022). No significant differences were detected by gender in the platelet counts between the unexposed and exposed subjects to the oil spill. Males in the oil spill-exposed group had significantly higher hemoglobin (15.1 1.5 vs 14.6 1.3 g/dL, P ¼ .022) and hematocrit (45.1 3.2 vs 42.9 5.4%, P ¼ .000) levels compared with the males in the unexposed group. BUN (mg per dL) was significantly decreased in males in the oil spill-exposed group compared with those in the unexposed group (13.4 3.4 vs 17.7 9.3, P ¼ .000).

The serum ALP levels (IU/L) were significantly elevated in male (76.9 21.0 vs 64.3 18.0, P ¼ .018) and female (71.3 23.1 vs 62.2 16.1, P ¼ .048) subjects in the oil spill-exposed group compared with the unexposed group. Similarly, the serum levels of AST and ALT (IU/L) were significantly elevated in both male and female subjects in the oil spill-exposed group compared with the unexposed group (Table 4).

The results in Table 5 represent the hematologic and hepatic changes according to the age group in oil spill-exposed subjects. The WBC (10^3 per mL) counts were significantly lower in the 20-29-year age group compared with the 30-39-year (6.2 1.6 vs 6.8 1.8, P ¼ .02) or over-40-year age group (6.2 1.6 vs 7.5 2.0, P ¼ .001). The WBC (10^3 per mL) counts were significantly lower in the 30-39-year age group compared with the over-40-year age group (6.8 1.8 vs 7.5 2.0, P ¼ .08). Although no differences were seen in beta-2-microglobulin (mg/L) levels between the 20-29-year and 30-39-year age groups, the 20-29-year age group had significantly lower levels of beta-2-microglobulin compared with the over-40-year age group (1.3 0.4 vs 1.4 0.3, P ¼ .02). Similarly, the 30-39-year age group had significantly lower levels of beta-2-microglobulin compared with the over-40-year age group (1.3 0.4 vs 1.4 0.3, P ¼ .05).

The principal somatic symptoms complained about by the oil spill clean-up workers are illustrated in Figure 2. Headache (77%) was the most frequently reported symptom, followed by shortness of breath (71%), skin rash (59%), cough (51%), dizzy spells (51%), fatigue (50%), painful joints (49%), night sweats (41%), and chest pain (38%).

The findings presented in Table 6 reveal the occurrence of somatic symptoms by sex among oil clean-up workers. Males had a lower frequency of headaches compared with females following exposure to the oil spill (75% vs 92%). Similarly, males experienced a lower frequency of painful joints (46% vs 69%), night sweats (38% vs 62%), ringing in the ears (32% vs 69%), fatigue (48% vs 62%), memory loss (20% vs 46%), and diarrhea (26% vs 38%) compared with

Table 3 Comparison of Hematologic and Hepatic Indices by Age Group between Subjects Unexposed and Exposed to the Oil Spill

Variable	Age Group	Unexposed‡	Exposed§	P Value
WBC (× 10^3 per mL)	<40 years	6.5 ± 1.9	6.6 ± 1.7	.461†
	≥40 years	6.5 ± 1.9	7.5 ± 2.0	.007**
Platelets (× 10^3 per mL)	<40 years	283.6 ± 67.3	251.8 ± 51.7	.006**
	≥40 years	249.5 ± 60.7	252.5 ± 52.4	.134†
Hemoglobin (g per dL)	<40 years	13.7 ± 1.8	14.9 ± 1.3	.000**
	≥40 years	13.6 ± 1.6	14.9 ± 1.4	.000**
Hematocrit (%)	<40 years	40.2 ± 5.9	44.7 ± 3.4	.000**
	≥40 years	41.0 ± 4.8	44.4 ± 3.4	.000**
BUN (mg per dL)	<40 years	12.1 ± 3.8	13.6 ± 3.3	.039*
	≥40 years	15.7 ± 5.6	14.5 ± 2.8	.389†
Creatinine (mg per dL)	<40 years	0.8 ± 0.2	1.1 ± 0.2	.000**
	≥40 years	0.9 ± 0.3	1.0 ± 0.2	.044*
ALP (IU per L)	<40 years	65.0 ± 21.7	74.9 ± 20.9	.026*
	≥40 years	67.5 ± 20.2	78.6 ± 22.0	.001**
AST (IU per L)	<40 years	20.9 ± 6.2	31.6 ± 9.0	.050*
	≥40 years	23.5 ± 4.7	30.0 ± 5.8	.048*
ALT (IU per L)	<40 years	22.9 ± 12.5	33.8 ± 22.9	.050*
	≥40 years	26.0 ± 13.8	40.1 ± 20.7	.054*

ALP = alkaline phosphatase; ALT = alanine amino transferase; AST = aspartate amino transferase; BUN = blood urea nitrogen; WBC = white blood cells.
*P = 0.05.
**P = 0.001.
†Did not reach statistical significance.
‡Unexposed <40 years: n = 28; unexposed ≥40 years: n = 102.
§Exposed <40 years: n = 73; exposed ≥40 years: n = 44.

females, after exposure to the oil spill. Conversely, compared with females, males had a higher frequency of skin rash (61% vs 46%), chronic cough (55% vs 23%), and poor appetite (26% vs 8%).

The findings in Table 7 reveal the frequency of somatic symptoms according to the age group among oil spill clean-up workers. Subjects in the age group of 20-29 years reported higher frequency of painful joints compared with

Table 4 Comparison of Hematologic and Hepatic Indices by Sex between Unexposed and Exposed Subjects to Oil Spill

Variable	Sex	Unexposed‡	Exposed§	P Value
WBC (× 10^3 per mL)	Male	6.7 ± 2.2	6.8 ± 1.8	.343†
	Female	6.4 ± 1.7	7.5 ± 2.2	.022*
Platelets (× 10^3 per mL)	Male	234.6 ± 71.5	248.7 ± 52.0	.094†
	Female	287.0 ± 74.5	278.7 ± 42.5	.348†
Hemoglobin (g per dL)	Male	14.6 ± 1.5	15.1 ± 1.3	.022*
	Female	13.1 ± 1.5	13.3 ± 0.9	.331†
Hematocrit (%)	Male	42.9 ± 5.4	45.1 ± 3.2	.000**
	Female	39.8 ± 4.5	40.5 ± 2.3	.293†
BUN (mg per dL)	Male	17.7 ± 9.3	13.4 ± 3.4	.000**
	Female	14.1 ± 5.1	13.0 ± 3.6	.228†
Creatinine (mg per dL)	Male	1.1 ± 0.3	1.1 ± 0.2	.381†
	Female	0.8 ± 0.2	0.8 ± 0.2	.202†
ALP (IU per L)	Male	64.3 ± 18.0	76.9 ± 21.0	.018*
	Female	62.2 ± 16.1	71.3 ± 23.1	.048*
AST (IU per L)	Male	21.6 ± 6.8	32.7 ± 7.5	.051*
	Female	18.6 ± 7.3	20.7 ± 5.0	.046*
ALT (IU per L)	Male	27.4 ± 13.5	38.2 ± 27.0	.054*
	Female	17.6 ± 12.0	26.1 ± 24.6	.039*

ALP = alkaline phosphatase; ALT = alanine amino transferase; AST = aspartate amino transferase; BUN = blood urea nitrogen; WBC = white blood cells.
*P = 0.05.
**P = 0.001.
†Did not reach statistical significance.
‡Male unexposed: n = 43; female unexposed: n = 87.
§Male exposed: n = 104; female exposed: n = 13.

Table 5 Hematologic and Hepatic Changes by Age Group among Oil Spill Clean-up Workers

Variable	Age Group 20-29 Years (n ¼ 31)	Age Group 30-39 Years (n ¼ 42)	Age Over 40 Years (n ¼ 44)	P Value‡	P Value§	P Value¶
WBC (¬ 10^3 per mL)	6.2 └ 1.6	6.8 └ 1.8	7.5 └ 2.0	.02*	.001**	.08†
Hemoglobin (g per dL)	15.0 └ 1.3	14.8 └ 1.3	14.9 └ 1.4	.5†	.3†	.3†
Hematocrit (%)	45.0 └ 3.5	44.4 └ 3.3	44.4 └ 3.4	.5†	.2†	.5†
Platelets (¬ 10^3 per mL)	245.2 └ 50.1	256.6 └ 55.8	252.5 └ 52.9	.1†	.3†	.4†
BUN (mg per dL)	13.1 └ 3.2	13.4 └ 3.7	13.7 └ 3.3	.1†	.2†	.5†
Creatinine (mg per dL)	1.0 └ 0.2	1.1 └ 0.2	0.97 └ 0.2	.03*	.2†	.00**
b-2-microglobulin (mg per L)	76.1 └ 21.4	74.1 └ 20.5	78.6 └ 22.1	.3†	.3†	.1†
ALP (IU per L)	29.1 └ 29.0	33.4 └ 31	30.0 └ 21.4	.4†	.4†	.4†
AST (IU per L)	32.7 └ 32.1	34.5 └ 18.5	37.6 └ 29.9	.5†	.3†	.2†
ALT (IU per L)	1.3 └ 0.3	1.3 └ 0.4	1.4 └ 0.3	.5†	.02*	.05*
Urinary Phenol (mg per L)	8.7 └ 9.5	7.6 └ 9.9	5.3 └ 5.9	.47†	.03*	.09†

ALP ¼ alkaline phosphatase; ALT ¼ alanine amino transferase; AST ¼ aspartate amino transferase; BUN ¼ blood urea nitrogen; WBC ¼ white blood cells.
*P ¼ 0.05.
**P ¼ 0.001.
†Did not reach statistical significance.
‡Differences between age group of 20-29 years and 30-39 years.
§Differences between age group of 20-29 years and │40 years.
¶Differences between age group of 30-39 years and │40 years.

those in the 30-39-year or over-40-year age groups (65% vs 38% vs 48%). Similarly, the frequency of night sweats was higher in the 20-29-year age group compared with those in the 30-39-year or over-40-year age groups (52% vs 36% vs 39%). Conversely, subjects in the age group of 20-29 years reported the lowest frequency of chronic cough compared with those in the 30-39-year or over-40-year age groups (39% vs 52% vs 59%). The occurrence of other symptoms appeared to be similar among the 3 age groups.

DISCUSSION

Despite numerous oil spills that have affected the coastal populations, there have been very few studies that evaluated the health effects of these oil spills. Therefore, in this retrospective study we evaluated the health effects of the Deepwater Horizon oil spill and dispersant use in a cohort of oil spill clean-up workers. To our knowledge, no previous study has explored the effects of the oil spill specifically assessing the hematological and hepatic functions in oil spill clean-up workers.

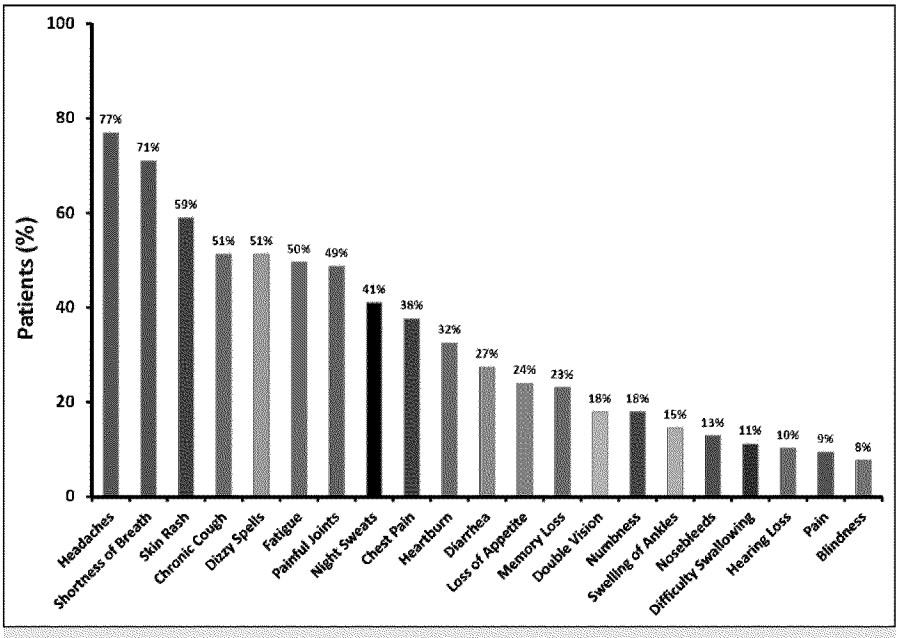

Figure 2 Major somatic symptoms experienced by the oil spill clean-up workers following exposure to oil spill and dispersant.

Table 6 Frequency of Somatic Symptoms by Sex among Oil Spill Clean-up Workers

Variable	Male (n ¼ 104)	Female (n ¼ 13)
Headaches	78 (75%)	12 (92%)
Shortness of breath	74 (71%)	9 (69%)
Skin rash	63 (61%)	6 (46%)
Chronic cough	57 (55%)	3 (23%)
Fatigue	50 (48%)	8 (62%)
Painful joints	48 (46%)	9 (69%)
Chest pain	40 (38%)	4 (31%)
Night sweats	40 (38%)	8 (62%)
Heartburn	33 (32%)	5 (38%)
Ringing in ears	33 (32%)	9 (69%)
Diarrhea	27 (26%)	5 (38%)
Poor appetite	27 (26%)	1 (8%)
Memory loss	21 (20%)	6 (46%)

The results of this study indicate that oil spill exposure appears to play a role in the development of hematologic and hepatic toxicity. While WBC levels remained similar, platelet levels were significantly decreased in oil spill-exposed subjects compared with the unexposed subjects. Similarly, the BUN and creatinine levels were significantly decreased in oil spill-exposed subjects compared with the unexposed subjects. Conversely, the hemoglobin and hematocrit levels were significantly increased among oil spill-exposed subjects compared with the unexposed subjects. Although the oil spill-exposed subjects had significant differences in various hematological and hepatic indices, our results indicate that exposure to the oil spill did not uniformly affect subjects who participated in oil spill clean-up activities.

The levels of ALP, AST, and ALT were increased among subjects exposed to the oil spill compared with those of the unexposed subjects. These serum enzyme levels have been considered indicators of hepatic damage. Phosphatases, amino transferases, and dehydrogenases play critical roles in biological processes. Specifically, these enzymes are involved in detoxification, metabolism, and biosynthesis of energetic macromolecules that are important for different essential functions. Alterations in the levels of these enzymes result in biochemical impairment and lesions in the tissue and cellular function. Thus, these enzymes are considered as specific markers for hepatic dysfunction and damage.[24] Similar increased levels of these enzymes have been reported in clinical and preclinical studies after benzene exposure.[25-27] The increase in the levels of these hepatic enzymes in the serum may be due to impairment in the function of hepatic tissues following exposure to toxic chemicals from an oil spill, such as benzene, thereby allowing liberation of these enzymes into the circulation. Furthermore, the serum ALP is often used as a marker to measure hepatic and biliary tract function.[28,29] The increase in the serum levels of ALP observed in this study suggest the possibility of biliary tract dysfunction in oil spill-exposed subjects.

Table 7 Frequency of Somatic Symptoms by Age Group among Oil Spill Clean-up Workers

Symptoms	Age Group 20-29 Years (n ¼ 31)	Age Group 30-39 Years (n ¼ 42)	Age Over 40 Years (n ¼ 44)
Headaches	25 (81%)	33 (79%)	32 (73%)
Painful joints	20 (65%)	16 (38%)	21 (48%)
Skin rash	19 (61%)	23 (55%)	27 (61%)
Shortness of breath	18 (58%)	34 (81%)	31 (70%)
Night sweats	16 (52%)	15 (36%)	17 (39%)
Fatigue	15 (48%)	20 (48%)	23 (52%)
Dizzy spells	15 (48%)	24 (57%)	21 (48%)
Chest pain	13 (42%)	20 (48%)	11 (25%)
Chronic cough	12 (39%)	22 (52%)	26 (59%)
Heartburn	11(35%)	13 (31%)	14 (32%)
Ringing in ears	11(35%)	13 (31%)	18 (41%)
Poor appetite	8 (26%)	11 (26%)	9 (20%)
Memory loss	8 (26%)	5 (12%)	14 (32%)
Diarrhea	7 (23%)	13 (31%)	12 (27%)

Traditionally, the excretion of phenol in the urine has been used as an index of benzene exposure, which is a principal component of crude oil.[30-32] In this study, we found that there were significant amounts of phenol (7.0 mg/L) in the urine of the oil spill-exposed subjects. However, in healthy subjects (not exposed to oil spill), only traceable or an undetectable amount of phenol is expected in the urine. Thus, these findings suggest that the subjects involved in the oil spill clean-up activity were inherently exposed to benzene.

The health complaints reported by those involved in the oil spill clean-up operations are consistent with the previously reported studies of other major oil spills.[6,7,9,33] However, the prevalence of symptoms appears to be higher in the present study compared with the earlier findings of other investigators. Lyons et al.[6] found that headache, shortness of breath, fatigue, cough, skin rash, and diarrhea occurred more frequently in subjects exposed to the Sea Empress oil spill. Similarly, Morita et al[33] observed headache, low back pain, leg pain, and jiggling of vision as major symptoms in subjects engaged in the clean-up activity of the Nakhodka oil spill. In the present study, the major symptoms observed were headache, shortness of breath, skin rash, cough, and fatigue in subjects involved in oil spill clean-up operations.

To understand the health consequences of oil spill exposure, a subgroup analysis was performed in subjects who participated in the oil spill clean-up activity. Specifically, the consequences of oil spill exposure were examined according to sex and age by subgroup analysis. Although the number of female clean-up workers was limited, we found no significant differences in the WBC count, BUN, beta-2-microglobulin, ALP, AST, ALT, and urinary phenol between the male and female subjects exposed to the oil spill. However, the platelet counts were significantly lower

in the male subjects compared with the female subjects exposed to the oil spill. Age appears to have minimal impact on various hematological and hepatic parameters in subjects exposed to the oil spill.

There were certain differences observed in the frequency of somatic symptoms between the male and female subjects involved in the oil spill clean-up activity. Compared with males, females experienced a higher rate of headaches, painful joints, fatigue, night sweats, and diarrhea following the exposure to the oil spill. Similarly, the age of the subjects appears to have a considerable impact on the frequency of somatic symptoms after the oil spill exposure.

Study Limitations

As with any cross-sectional study and most studies of oil spill disasters, our results should be considered in light of the following limitations. Foremost among these limitations is our lack of predisaster health data on the subjects involved in the clean-up activities. In addition, the demographic characteristics such as sex and age were significantly different between the exposed and unexposed groups and may have impacted the observed findings. Specifically, the median age of unexposed subjects was significantly higher when compared with the exposed group. Furthermore, data on somatic symptoms were self-reported by the oil spill-exposed group and may have influenced the findings of the study. Also, because the clinical outcomes were measured at one time point after the oil spill exposure, it was difficult to infer a causality using such a study design. Moreover, the effect of the oil dispersant may have contributed to the current findings. The COREXIT used as a dispersant is currently banned in the United Kingdom because of its potential health risks to clean-up workers.[5] Nonetheless, this study has shed light on the health effects of oil spill exposure and dispersant use in subjects participating in the clean-up activities.

CONCLUSION

The results of this study indicate that human exposure to the oil spill has a potential to induce both hematological and hepatic toxicity. The hematological alterations include depletion of platelets, decreased BUN, and increased creatinine in subjects exposed to a crude oil spill. In addition, headache, shortness of breath, skin rash, cough, fatigue, painful joints, and chest pain occurred more frequently in oil spill clean-up workers. However, additional long-term follow-up studies are required to understand the clinical significance of the oil spill exposure.

ACKNOWLEDGMENT

The authors are thankful to Lingamanaidu V. Ravichandran, PhD, for critical review of the manuscript and for providing medical writing support and to Pradheeth Reddy for his work in editing and proofreading the manuscript.

References

1. Urum K, Grigson S, Pekdemir T, McMenamy S. A comparison of the efficiency of different surfactants for removal of crude oil from contaminated soils. Chemosphere. 2006;62:1403-1410.
2. Solomon GM, Janssen S. Health effects of the Gulf oil spill. JAMA. 2010;304:1118-1119.
3. Merhi ZO. Gulf Coast oil disaster: impact on human reproduction. Fertil Steril. 2010;94:1575-1577.
4. Goldstein BD, Osofsky HJ, Lichtveld MY. The Gulf oil spill. N Engl J Med. 2011;364:1334-1348.
5. Diaz JH. The legacy of the Gulf oil spill: analyzing acute public health effects and predicting chronic ones in Louisiana. Am J Disaster Med. 2011;6:5-22.
6. Lyons RA, Temple JM, Evans D, Fone DL, Palmer SR. Acute health effects of the Sea Empress oil spill. J Epidemiol Community Health. 1999;53:306-310.
7. Cheong HK, Ha M, Lee JS, et al. Hebei spirit oil spill exposure and subjective symptoms in residents participating in clean-up activities. Environ Health Toxicol. 2011;26:e2011007.
8. Carrasco JM, Lope V, Perez-Gomez B, et al. Association between health information, use of protective devices and occurrence of acute health problems in the Prestige oil spill clean-up in Asturias and Cantabria (Spain): a cross-sectional study. BMC Public Health. 2006;6:1.
9. Janjua NZ, Kadir MM, Lutfi S, Tipre M, Sathiakumar N. Tasman spirit oil spill in Pakistan: Research response and lessons learned. Am J Ind Med. 2012;56:124-131.
10. Gill DA, Picou JS, Ritchie LA. The Exxon Valdez and BP oil spills: a comparison of initial social and psychological impacts. Am Behav Sci. 2012;56:3-23.
11. Repanich J. The Deepwater Horizon spill by the numbers. Available at: http://www.popularmechanics.com/science/energy/coal-oil-gas/bp-oil-spill-statistics. Accessed May 22, 2013.
12. Biello D. Is using dispersants on the BP Gulf oil spill fighting pollution with pollution? Sci Am. 2010;June 18:22.
13. Sandler DP, Kwok RK, Engel LS, Parks C, London SJ, Miller AK. GuLF Study: gulf long-term follow-up study. Available at: http://www.niehs.nih.gov/research/programs/gulfspill/gulfstudy/backgrounddocuments/gulf_study_protocol_7092012.pdf. Accessed May 22, 2013.
14. Aguilera F, Mendez J, Pasaro E, Laffon B. Review on the effects of exposure to spilled oils on human health. J Appl Toxicol. 2010;30:291-301
15. Bosch X. Exposure to oil spill has detrimental effect on clean-up workers' health. Lancet. 2003;361:147.
16. Murray JS. The effects of the gulf oil spill on children. J Spec Pediatr Nurs. 2011;16:70-74.
17. Costantini AS, Benvenuti A, Vineis P, et al. Risk of leukemia and multiple myeloma associated with exposure to benzene and other organic solvents: evidence from the Italian Multicenter Case-control study. Am J Ind Med. 2008;51:803-811.
18. Travis LB, Li CY, Zhang ZN, et al. Hematopoietic malignancies and related disorders among benzene-exposed workers in China. Leuk Lymphoma. 1994;14:91-102.
19. Hayes RB, Yin S, Rothman N, et al. Benzene and lymphohematopoietic malignancies in China. J Toxicol Environ Health A. 2000;61:419-432.
20. Bloemen LJ, Youk A, Bradley TD, Bodner KM, Marsh G. Lymphohaematopoietic cancer risk among chemical workers exposed to benzene. Occup Environ Med. 2004;61:270-274.
21. Khalade A, Jaakkola MS, Pukkala E, Jaakkola JJ. Exposure to benzene at work and the risk of leukemia: a systematic review and meta-analysis. Environ Health. 2010;9:31.
22. McHale CM, Zhang L, Smith MT. Current understanding of the mechanism of benzene-induced leukemia in humans: implications for risk assessment. Carcinogenesis. 2012;33:240-252.
23. Reardon S. Gulf oil spill. Ten months after Deepwater Horizon, picking up the remnants of health data. Science. 2011;331:1252.
24. Khan IA, Reddy BV, Mahboob M, Rahman MF, Jamil K. Effects of phosphorothionate on the reproductive system of male rats. J Environ Sci Health Part B. 2001;36:445-456.

25. Shamy MY, el Gazzar RM, el Sayed MA, Attia AM. Study of some biochemical changes among workers occupationally exposed to phenol, alone or in combination with other organic solvents. Ind Health. 1994;32:207-214.
26. Dere E, Ari F. Effect of Benzene on liver functions in rats (Rattus norvegicus). Environ Monit Assess. 2009;154:23-27.
27. Dede EB, Kagbo HD. A study on the acute toxicological effect of commercial diesel fuel in Nigeria in rats (Ratus ratus) using hematological parameters. J Appl Sci Environ Manage. 2002;6:84-86.
28. Polin SG, Spellberg MA, Teitelman L, Okumura M. The origin of elevation of serum alkaline phosphatase in hepatic disease. An experimental study. Gastroenterology. 1962;42:431-438.
29. Sebesta DG, Bradshaw FJ, Prockop DJ. Source of the elevated serum alkaline phosphatase activity in biliary obstruction: studies utilizing isolated liver perfusion. Gastroenterology. 1964;47:166-170.
30. Docter HJ, Zielhuis RL. Phenol excretion as a measure of benzene exposure. Ann Occup Hyg. 1967;10:317-326.
31. Inoue O, Seiji K, Kasahara M, et al. Quantitative relation of urinary phenol levels to breathzone benzene concentrations: a factory survey. Br J Ind Med. 1986;43:692-697.
32. McDonald TA, Holland NT, Skibola C, Duramad P, Smith MT. Hypothesis: phenol and hydroquinone derived mainly from diet and gastrointestinal flora activity are causal factors in leukemia. Leukemia. 2001;15:10-20.
33. Morita A, Kusaka Y, Deguchi Y, et al. Acute health problems among the people engaged in the cleanup of the Nakhodka oil spill. Environ Res. 1999;81:185-194.

CLINICAL RESEARCH STUDY

THE AMERICAN JOURNAL of MEDICINE

Health Risks Associated with Crude Oil Spill Exposure

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ABSTRACT

BACKGROUND: Human exposure to crude oil spills is associated with multiple adverse health effects including hematopoietic, hepatic, renal, and pulmonary abnormalities. The purpose of this study was to assess the hematological and liver function indices among the subjects participating in the Gulf oil spill cleanup operations in comparison with the standardized normal range reference values.

METHODS: Using medical charts, clinical data (including white blood cell [WBC] count, platelet count, hemoglobin, hematocrit, blood urea nitrogen [BUN] creatinine, alkaline phosphatase [ALP], aspartate amino transferase [AST], alanine amino transferase [ALT], and urinary phenol) were gathered for the subjects who were exposed to the Gulf oil spill and analyzed.

RESULTS: A total of 117 subjects exposed to the oil spill were included. Over 77% of subjects had WBC counts in the midrange (6-10 10^3 per mL), while none of the subjects had the upper limit of the normal range (11 10^3 per mL). A similar pattern was seen in the platelet counts and BUN levels among the oil spill-exposed subjects. Conversely, over 70% of the subjects had creatinine levels toward the upper limit of the normal range and 23% of subjects had creatinine levels above the upper limit of the normal range (> 1.3 mg per dL). Similarly, hemoglobin and hematocrit levels were toward the upper limit of normal in more than two thirds of the subjects. AST and ALT levels above the upper limit of normal range (> 40 IU per L) were seen in 15% and 31% of subjects, respectively. Over 80% of subjects had urinary phenol levels higher than detectable levels (2 mg per L).

CONCLUSION: The results of this study support our earlier study findings in which we found that people who participated in oil spill cleanup activities are at risk of developing alterations in hematological profile and liver function.

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KEYWORDS: Chemical exposure; Health impact; Hematological toxicity; Hepatotoxicity; Urinary phenol

Crude oil spills are tragic environmental disasters that can affect human health through exposure to inherent toxic chemicals including para-phenols and volatile benzene.[1,2] Human exposure to crude oil spills can lead to deleterious effects on many biological systems, including changes in the hematologic, hepatic, respiratory, renal, and neurological functions.[3,4] Recently, we investigated the adverse health effects of the Deepwater Horizon oil spill exposure in subjects participating in oil spill cleanup activities along the coast of Louisiana.[5] Specifically, we assessed the hematologic and hepatic markers in a cohort of oil spill cleanup workers and the clinical findings were compared with a group of unexposed (control) subjects. The findings of the study revealed significant differences in blood profiles and liver enzymes between the oil spill exposed and unexposed cohorts, indicating health risks among subjects who participated in the oil spill cleanup operation.[5]

Because the most reported clinical biomarker findings for both hematological and liver function indices fell within the normal range values for exposed and unexposed groups, concerns were raised about the study design and interpretation of the study findings.[6,7] This prompted us to carry out an additional analysis of clinical data that has been recently reported.[5] In the analysis, we assessed the blood profile and liver function data in comparison with the standard normal range values among the subjects who participated in the

Funding: None.

Conflict of Interest: None.

Authorship: Both authors listed in the manuscript had access to the data and a role in preparing the manuscript.

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http://dx.doi.org/10.1016/j.amjmed.2014.04.035

Gulf oil spill cleanup operation. The resultant outcomes are summarized to further facilitate the understanding of the health impact of oil spill exposure among subjects observed previously.

MATERIALS AND METHODS

Subjects

The details of identification of the subjects exposed to the oil spill were described in our previous manuscript.[5] Briefly, the subjects exposed to the oil spill were identified as participants in the oil spill cleanup activities along the coast of Louisiana. Medical charts were reviewed by an experienced physician. Clinical data such as white blood cell (WBC) counts, platelet counts, hemoglobin, hematocrit, blood urea nitrogen (BUN), creatinine, alkaline phosphatase (ALP), aspartate aminotransferase (AST), and alanine aminotransferase (ALT) levels were collected and evaluated. In addition, data on urinary phenol was also assessed as a benzene metabolite in the oil spill-exposed subjects.

CLINICAL SIGNIFICANCE

- Crude oil spills are tragic environmental disasters that have high potential to affect human health.
- The results of this study indicate that humans exposed to crude oil spill have a risk of developing hepatic or blood-related disorders.
- The hematological and hepatic alterations include altered platelet counts, serum creatinine, urinary phenol, and liver function enzymes.

Analysis

As shown in Table 1, the average reference values were derived from the normal range values for various indices of hematological and liver functions and the clinical data were processed. In addition, the normal range values were divided into various categories and the clinical data were analyzed. The variables included WBC and platelet counts, as well as the levels of hemoglobin, hematocrit, creatinine, BUN, ALP, AST, ALT, and urinary phenol.

Table 1 Normal Range Values for Various Indices

Parameter	Normal Range Value	Optimal (Average) Normal Value
WBC (10^3 per mL)	4.0-10.5	7.25
Platelets (10^3 per mL)	140-415	276
Hemoglobin (g per dL)	12.6-17.7	15
Hematocrit (%)	37.5-51.0	44
BUN (mg per dL)	6-24	15
Creatinine (mg per dL)	0.76-1.27	1.0
ALP (IU per L)	25-150	88
AST (IU per L)	0-40	20
ALT (IU per L)	0-40	20
Beta-2 macroglobulin (mg per L)	0.6-2.4	1.5
Urinary phenol (mg per L)	0-trace	2

ALP ¼ alkaline phosphatase; ALT ¼ alanine aminotransferase; AST ¼ aspartate aminotransferase; BUN ¼ blood urea nitrogen; IU per L ¼ International Units per liter; WBC ¼ White blood cells.

RESULTS

This study included a total of 117 subjects who were involved in the cleanup operations of the oil spill. Of the 117 subjects, 104 (89%) were male and 13 (11%) were female. The median age of the subjects was 34.0 (18-63) years.

The findings presented in the Figure (panel A) indicate the outcomes of WBC, platelet counts, and BUN levels assessment in the oil spill-exposed subjects. Based on the normal range values, WBC count was averaged at 7.2 (10^3 per mL) and the total number of subjects who had WBC counts below and above the average value of normal range was determined. Of the 117 subjects, 71 (61%) had WBC counts below the average value of normal range. Similarly, 79 (68%) of the subjects had platelet counts below the average value of normal range when 276 (10^3 per mL) was considered as the average value of normal range. Assessment of BUN levels also indicated that 68% of the subjects had below the average value of normal range when the average value of normal range of BUN levels was considered to be at 15 mg per dL.

The results shown in the Figure (panel B) depict the outcomes of serum creatinine, hemoglobin, hematocrit, and urinary phenol levels assessment in the oil spill-exposed subjects. Based on the normal range values, serum creatinine levels averaged 1.0 mg per dL and the total number of subjects who had creatinine levels above and below the average value of normal range was determined. Of the 117 subjects, 57 (49%) had serum creatinine levels above the average. Similarly, 76 (65%) of the subjects had hemoglobin levels above the average when 15 mg per dL was considered as the average value of normal range. Assessment of hematocrit levels indicated that 65% of the subjects had levels above the average value of normal range when the average value of normal range of hematocrit levels was considered to be at 44 mg per dL. Similarly, 93 (79%) of the subjects had urinary phenol levels above the average value of the normal range when 2.0 mg per L was considered as the average value of the normal range.

The findings presented in the Figure (panel C) reveal the outcomes of liver function enzymes such as AST, ALT, and ALP levels assessment in the serum of oil spill-exposed subjects. Based on the normal range values, AST and ALT levels in the serum were averaged at 20 IU per L, and the total number of subjects who had AST or ALT levels above and below the average was determined. Of the 117 subjects, 84 (72%) had serum AST levels above the average of the normal range. Similarly, 89 (76%) subjects had serum

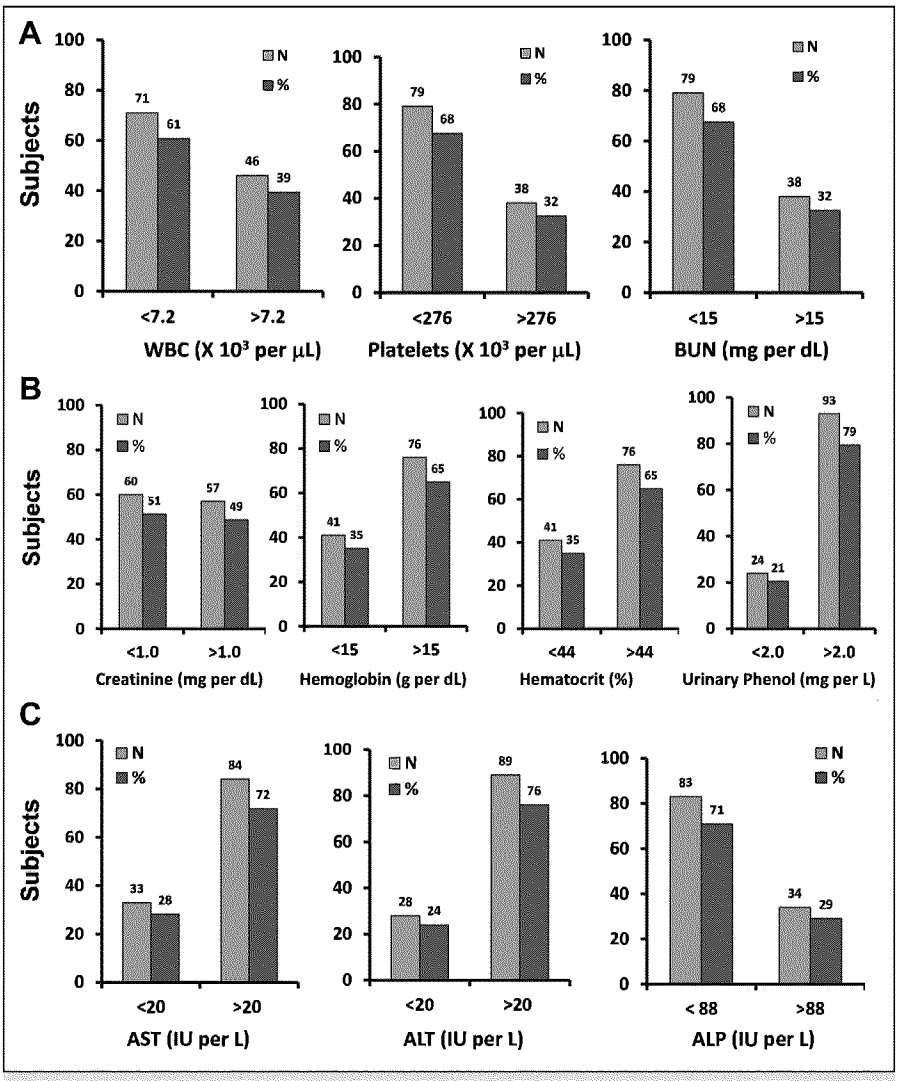

Figure Changes in biological markers in subjects involved in the Gulf oil spill cleanup operation. (A) Changes in white blood cell (WBC) counts, platelet counts, and blood urea nitrogen (BUN) levels. (B) Changes in serum creatinine, hemoglobin, hematocrit, and urinary phenol levels. (C) Changes in liver function enzymes such as aspartate aminotransferase (AST), alanine aminotransferase (ALT), and alkaline phosphatase (ALP) levels.

ALT levels above the average of the normal range. Conversely, only 29% of the subjects experienced serum ALP levels above the average when 88 IU per L was considered as the average value of the normal range.

The outcomes were compared with the normal range reference values that were grouped into lower-, middle-, and upper-limit ranges in addition to the above-normal range. The results presented in Tables 2 and 3 indicate the outcomes on

Table 2 Blood Profile with Various Indices in Subjects (n ¼117) Involved in Oil Spill Cleanup Activity

WBC (10^3 per mL)		Platelets (10^3 per mL)		BUN (mg per dL)	
Count	n	Count	n	Level	n
<5.0	14 (12.0%)	<200	23 (19.7%)	<11.0	33 (28.2%)
6.0-<8.0	50 (42.7%)	200-<250	36 (30.8%)	11.0-<15.0	46 (39.3%)
8.0-<10.0	40 (34.2%)	250-<300	35 (29.9%)	15.0-<20.0	32 (27.4%)
10.0 -<11.0	13 (11.1%)	300-<400	23 (19.7%)	20.0-<24.0	6 (5.1%)
┘ 11.0	0 (0.0%)	┘ 400	0 (0.0%)	┘ 24.0	0 (0.0%)

BUN ¼ blood urea nitrogen; WBC ¼ white blood cells.

Table 3 Blood Profile and Urinary Phenol in Subjects (n ¼117) Involved in Oil Spill Cleanup Activity

Creatinine (mg per dL)		Hemoglobin (g per dL)		Hematocrit (%)		Urinary Phenol (mg per L)	
Level	n	Level	n	Level	n	Level	n
<0.80	11 (9.4%)	<13.0	21 (17.9%)	<40	11 (9.4%)	0	10 (8.5%)
0.8-<0.9	22 (18.8%)	13.0-<15.0	20 (17.1%)	40-<45	42 (35.9%)	<2.0	13 (11.0%)
0.9-<1.0	27 (23.1%)	15.0-<16.0	60 (51.3%)	45-<50	56 (47.9%)	2.0-<5.0	38 (32.5%)
1.0-<1.3	30 (25.6%)	┘ 16.0	16 (13.7%)	┘ 50	8 (6.8%)	5.0-<10.0	31 (26.5%)
┘ 1.3	27 (23.1%)	N/A	N/A	N/A	N/A	┘ 10.0	25 (21.4%)

blood profiles and urinary phenol levels in subjects involved in the oil spill cleanup operations. Over 77% of the subjects had WBC counts in the mid range (6-10 10^3 per mL), while none of the subjects had upper limits of the normal range (11 10^3 per mL). A similar pattern was seen in the platelet counts and BUN levels among the oil spill-exposed subjects. However, nearly one third of subjects had BUN levels less than the lower limit (11 mg per dL) of the normal range. Conversely, over 70% of the subjects had creatinine levels toward the upper limit of the normal range and 23% of subjects had creatinine levels above the upper limits of the normal range (>1.3 mg per dL, Table 3). Similarly, hemoglobin levels were raised toward the upper limit of the normal range (>15 g per dL) in more than two thirds of the subjects. Hematocrit levels were also in the upper limit of the normal range in over 50% of the subjects. Over 80% of the subjects had urinary phenol levels more than detectable levels (2 mg per L) and 21% of the subjects had urinary phenol levels >10 mg per L (Table 3).

The findings presented in Table 4 show the levels of the hepatic enzymes such as AST, ALT, and ALP in the serum of the subjects involved in oil spill cleanup operations. AST levels above the normal range (>40 IU per L) were seen in approximately 15% of the subjects. Similarly, ALT levels above the normal range (>40 IU per L) values were seen in approximately 31% of the subjects. Conversely, none of the subjects had ALP levels above the normal range (>150 IU per L) values.

DISCUSSION

In a previous study, we evaluated the health effects of the Deepwater Horizon oil spill exposure in a cohort of oil spill cleanup workers.[5] The study found that the blood profiles and hepatic enzymes were significantly altered in subjects exposed to the oil spill compared with the unexposed subjects. Because the standard reference values have a wide range for the indices, a follow-up study was undertaken to assess the outcomes with respect to the standard reference values. Specifically, the outcomes were compared with the average value of the normal range values of each parameter. Further, the outcomes were compared with the normal range reference values that were grouped into lower-, middle-, and upper-limit ranges in addition to the above-normal range.

The findings of the study indicate that subjects who participated in the oil spill cleanup operations experienced changes in their blood profiles and hepatic enzymes. The study findings indicated that about 60%-70% of subjects exposed to the oil spill had WBC and platelet counts as well as BUN levels below the average normal reference value. Conversely, 50%-65% of subjects exposed to the oil spill had creatinine, hemoglobin, and hematocrit levels above the average normal reference value. Urinary phenol levels were also found to be above the average normal reference value in approximately 80% of oil spill-exposed subjects. Similarly, liver function enzymes such as AST and ALT were found to be above the average normal reference value in over 70% of oil spill-exposed subjects. Thus, these findings support our earlier study findings in which we found that subjects who participated in the oil spill cleanup activity are at an increased risk of developing hematological and liver function abnormalities.[5]

The abnormality of various biological markers in subjects who are involved in oil spill cleanup operations is further evidenced by the changes in various indices including creatinine, hemoglobin, hematocrit, liver function

Table 4 Liver Function Enzymes in the Serum of Subjects (n ¼117) Involved in Oil Spill Cleanup Activity

AST (IU per L)		ALT (IU per L)		ALP (IU per L)	
Level	n	Level	n	Level	n
<20	33 (28.2%)	<20	28 (23 .9%)	<60	27 (23.1%)
20-<30	49 (41.9%)	20-<30	38 (32.5%)	60-<80	48 (41.0%)
30-<40	18 (15.4%)	30-<40	15 (12.8%)	80-<100	23 (19.7%)
┘ 40	17 (14.5%)	┘ 40	36 (30.8%)	┘ 100	19 (16.2%)

ALP ¼ alkaline phosphatase; ALT ¼ alanine aminotransferase; AST ¼ aspartate aminotransferase.

enzymes (AST and ALT), and urinary phenol. The results show that over 15% of subjects experienced some of these biomarker indices over the upper limit of the normal reference range, and over 50% of subjects had the indices toward the upper limit.

Overall, the findings indicate that the subjects were not affected uniformly by the oil spill exposure. This could be due to the differences in duration and intensity of exposure, protective equipment, preexisting health conditions, use of medications, and the differential immunity to the toxic chemicals of the oil spill. In addition, subjects' age may have contributed to the variations in findings observed in the study. Nonetheless, the findings of the study suggest that subjects involved in oil spill cleanup operation are at an increased risk of developing health-related sequelae and support our earlier findings reported in the previous article.[5]

ACKNOWLEDGMENT

The authors are thankful to Lingamanaidu V. Ravichandran, PhD, for critical review of the manuscript and for providing medical writing support, and to Pradheeth Reddy for his work in editing and proofreading the manuscript.

References

1. Solomon GM, Janssen S. Health effects of the Gulf oil spill. JAMA. 2010;304:1118-1119.
2. Merhi ZO. Gulf Coast oil disaster: impact on human reproduction. Fertil Steril. 2010;94:1575-1577.
3. Goldstein BD, Osofsky HJ, Lichtveld MY. The Gulf oil spill. N Engl J Med. 2011;364:1334-1348.
4. Diaz JH. The legacy of the Gulf oil spill: analyzing acute public health effects and predicting chronic ones in Louisiana. Am J Disaster Med. 2011;6:5-22.
5. D'Andrea MA, Reddy GK. Health consequences among subjects involved in Gulf oil spill clean-up activities. Am J Med. 2013;126:966-974.
6. Wickliffe JK, Wilson MJ, Lichtveld MY. Major Concerns About Study Design and Clinical Biomarker Interpretation. Am J Med. 2014;127(9):e21-e22.
7. Piacentino J, Silver S, Bernard B, DeBord DG, Funk R, Decker J. Study methodology prevents interpretation of findings in workers involved in Gulf oil spill cleanup activities. Am J Med. 2014;127(9):e25-e26.

THE AMERICAN JOURNAL of MEDICINE

Major Concerns About Study Design and Clinical Biomarker Interpretation

To the Editor:

We are writing this letter in response to the recent article in The American Journal of Medicine by D'Andrea and Reddy.[1] Our primary concerns are with their convenience sampling and case-control mismatching, and the authors' conclusions about the clinical results. The clinical results indicate that both groups in their study, unexposed and exposed, have clinical biomarker values for both liver function and blood profiles that essentially fall within the normal ranges for healthy adult individuals (Table).[2-4] Results from clinical laboratories tend to vary only slightly for such routine and standardized tests. Statistical differences among such unmatched groups do not necessarily equate to biologically meaningful differences. Their results should include an explicit reference and comparison to well-documented clinical ranges for normal, healthy adult individuals. An alternative conclusion about their clinical results is that while these age- and sex-biased convenience samples differ from one another statistically, neither group actually differs from accepted values for normal, healthy adult individuals. This equally valid conclusion might well serve to comfort the patients under evaluation in that, as individuals, none appear to have any clinical indications of abnormal liver or hematological function. We hold that this should have been explicitly discussed, and we are concerned that the conclusion reported by the authors may well exacerbate health problems through psychosocial and behavioral means in those studied, as well as others that participated in the cleanup process. How, then, would the authors and physicians in general interpret the relevance of their statistical findings, especially when comparing a predominantly male group of younger cases to a predominantly female group of older controls, and how would they communicate these biomarker results to their patients?

Beyond biases in interpreting their biomarker data, no attempt at reconstructing exposure also is a problem. We acknowledge the difficulty and uncertainty in retrospectively reconstructing exposures, but some level of exposure assessment beyond the label of cleanup worker is necessary.

Funding: Funding was received from the National Institutes of Health/National Institute of Environmental Health Sciences grant 5 U19 ES020677 03, the Baton Rouge Area Foundation, and the Gulf Region Health Outreach Program.

Conflict of Interest: None.

Authorship: All authors had access and participated in drafting this manuscript.

Table Clinical Ranges for Normal Healthy Adult Female and Male Subjects for the Liver Function and Hematological End Points under Study

	Normal Clinical Ranges				Unexposed		Exposed	
Clinical biomarker	Low*	High*	Low†	High†	Low‡	High‡	Low§	High§
Platelets, mcL	150,000	400,000	150,000	450,000	192,300	346,900	200,300	303,900
Hemoglobin, gm/dL (females)	12.1	15.1	12	15.5	12	15.2		
Hemoglobin, gm/dL (males)	13.8	17.2	13.5	17.5			13.6	16.2
Hematocrit, % (females)	36.1	44.3	34.9	44.5	35.8	45.8		
Hematocrit, % (males)	40.7	50.3	38.8	50			41.2	48
BUN, mg/dL	6	20	7	20	8.3	22.5	10	16.8
Creatinine, mg/dL (females)	0.6	1.1			0.6	1.2		
Creatinine, mg/dL (males)	0.7	1.3					0.8	1.2
ALP, IU/L	44	147	45	115	34.3	88.1	55	97.6
AST, IU/L	10	34	8	48	7.7	37.9	4.7	57.3
ALT, IU/L	10	40	7	55	2.8	56.8	8.2	61.4
Urinary phenol	5.9	13.1						

ALP ¼ alkaline phosphatase; ALT ¼ alanine aminotransferase; AST ¼ aspartate aminotransferase; BUN ¼ blood urea nitrogen.
*From the National Library of Medicine Web site.[2]
†From the Mayo Clinic Web site.[3]
‡From the Agency for Toxic Substances Disease Registry Web site.[4]
§Values from Table 2 in D'Andrea and Reddy (2013).[1]

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The authors do not include critical elements of an exposure assessment such as individual environmental and occupational health histories. Environmental and occupational histories would shed light on relevant concomitant non-oil-spill exposures, pre-, during-, and postspill activities and exposures that need to be considered, and any use of personal protective equipment during cleanup activities that should be accounted for. In short, these authors do not appear to have followed acceptable clinical practice as established by members of the Association of Occupational and Environmental Health Clinics.

We do commend these authors for their focus on the health of oil spill cleanup workers. Although they do describe, in part, some limitations of their study, the obvious influences these have on the results of their statistical testing and ultimately, on their biased conclusions, are problematic. These problems affect the validity of their conclusions and indirectly threaten to further stress our Gulf Coast communities.

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http://dx.doi.org/10.1016/j.amjmed.2013.10.025

References

1. D'Andrea MA, Reddy GK. Health consequences among subjects involved in Gulf oil spill clean-up activities. Am J Med. 2013;126(11):966-974.
2. National Library of Medicine. MedLine Plus. Available at: http://www.nlm.nih.gov/medlineplus/. Accessed September 18, 2013.
3. Mayo Clinic. Available at: http://www.mayoclinic.com. Accessed September 18, 2013.
4. Agency for Toxic Substances Disease Registry, Centers for Disease Control and Prevention. Available at: http://www.atsdr.cdc.gov/toxprofiles/index.asp. Accessed September 18, 2013.

To: Wilson, Gregory[Wilson.Gregory@epa.gov]
From: Principe, Vanessa
Sent: Wed 6/15/2016 5:53:23 PM
Subject: BSEE Response System Planning Calculators
EDSP_dispersants-Man.pdf
EBSP_insituBurn-Man.pdf
ERSP_mechRecovery-Man.pdf

From Lori's presentation, the link she said was to be updated. Attached the downloaded mechanical, ISB and dispersant manuals. Didn't realize they had a system potential calculator for dispersants, perhaps you did. Curious to see what the planning assumptions are.

Response System Planning Calculators

http://www.bsee.gov/About-BSEE/Divisions/OSPD/Response-System-Planning-Calculators/

Response System Planning Calculators

BSEE, in collaboration with Genwest, Inc., developed a set of response system planning calculators for mechanical recovery (ERSP), in situ burning (EBSP), and surface-applied dispersants (EDSP). These products are now publicly available and can be downloaded below. The use of these tools is currently not required for the submission of an oil spill response plan (OSRP); however, BSEE believes these calculators are best practices that offer significant improvements in estimating the capabilities of oil spill response systems. BSEE encourages plan holders to consider using these tools in the preparation and submission of OSRPs. BSEE may use these calculators as an additional evaluation tool when reviewing OSRPs for compliance with existing regulations.

For Mechanical Recovery:
Estimated Recovery System Potential (ERSP) Calculator
ERSP User Manual

For *In Situ* Burning:
Estimated Burning System Potential (EBSP) Calculator
EBSP User Manual

For Surface-Applied Dispersants:

Estimated Dispersant System Potential (EDSP) Calculator
EDSP User Manual
Instructions for Adding the OSRL B727 Jet Aircraft to the EDSP Calculator

The ERSP, EBSP, and EDSP Calculators are intended as planning tools for estimating the potential of different oil spill response systems to mitigate (recover, burn or disperse) discharged oil relative to one another. These planning tools are NOT intended to be used as models for calculating system performance during an actual oil spill, which is affected by many additional factors not included in these Calculators.

BSEE and Genwest also developed an additional product called the Recovery System Evaluation Tool (ReSET), designed to aid plan holders and OSROs in understanding how changes made to a mechanical recovery system will result in different outcomes in the ERSP Calculator.

Recovery System Evaluation Tool (ReSET)
ReSET User Manual

If you have any additional questions or feedback regarding these planning tools, please contact either the BSEE OSPD Section in your area, or John Caplis, at BSEE Headquarters in Sterling, VA, at 703-787-1364 or **john.caplis@bsee.gov**.

Effective Daily Recovery Capacity (EDRC) Project

http://www.bsee.gov/Technology-and-Research/Oil-Spill-Response-Research/Projects/Project-673/

Estimated Dispersant System Potential (EDSP) Calculator User Manual

The ERSP, EBSP, and EDSP Calculators are intended as planning tools for estimating the potential of different oil spill response systems to mitigate (recover, burn or disperse) discharged oil relative to one another. These planning tools are NOT intended to be used as models for calculating system performance during an actual oil spill, which is affected by many factors such as the distribution of oil on the water surface, oil weathering, and other ambient on-scene conditions which are not included in these Calculators.

Prepared by BSEE and Genwest Systems, Inc.

February 2016

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1. Introduction

Dispersants are fluid chemicals that bond to oil molecules and separate them from water molecules, thus breaking up the oil. The result is tiny oil droplets that can biodegradeand disperse more quickly than a mass of oil.

The Estimated Dispersant System Potential (EDSP) Calculator developed in this project is Revision 1 of the Dispersant Mission Planner 2 (DMP2). The basic dispersant application algorithms remain the same. The programming environment used for the DMP2 is no longer supported. A document highlighting the major differences between the DMP2 and the EDSP Calculator can be found at www.genwest.com/genwest_publications.

The Calculator accounts for the performance of an aircraft or vessel dispersant application system as it sprays dispersant on an oil slick. The Calculator generates an "Estimated Dispersant System Potential" (EDSP) value for two Operating Periods following the discharge of an oil spill. The Calculator takes into account the time necessary to mobilize the system and the time needed to cascade the system from its home base to the incident staging site, if necessary. There is separate logic for aircraft platforms and vessel platforms that determines what is displayed for the two operating periods.

The EDSP Calculator is an HTML file, **EDSP-160225.html**, which runs in the following web browser versions:

PC (Win-XP, Win-7 and Win-8 environments)
- IE - 9, 10, 11 or greater
- Chrome 38 or greater
- Firefox 31 or greater
- Safari 5.1 or greater

Mac (OS 10.6 and greater)
- Safari 8.0 or greater
- Chrome 38 or greater
- Firefox 35 or greater

You can download the EDSP Calculator from the BSEE website. The "build" or version date of the calculator appears after the calculator name in the format – yymmdd.html (see Figure 7). This date stamp can verify that you are working with the latest version of the calculator.

This User's Guide provides guidance to the user on the various data inputs that must be entered into the EDSP Calculator regarding the dispersant application configuration being evaluated, and further explains the Calculator results.

The EDSP Calculator was developed by Genwest Systems, Inc. under GSA Contract GS-00F-0002W, BSEE Order No. E12-PD-00012 in consultation with BSEE and the United States Coast Guard.

1a. The EDSP Calculator's Purpose and Intended Use

The EDSP Calculator is primarily a planning tool for estimating the potential for dispersant application on spilled oil by defined dispersant application systems. It estimates the potential of a single system and does not address the total potential or number of systems that would be necessary for a plan. You can use the Calculator to evaluate the potential of a dispersant application system for different spill planning situations

- **Estimated Dispersant System Potential (EDSP),** a regulatory measure of a dispersant system's potential ability to treat oil.

- **Effective Daily Application Capacity (EDAC)**, a regulatory measure of how much dispersant can be applied by a dispersant application platform that is used in conjunction with U.S. Coast Guard regulated oil spill response plans. The EDAC assumptions and scenario can be found in Section 2b.

- **Operational spill planning**, where users are not constrained to EDAC values and assumptions.

To use the Calculator, you enter information about an aircraft or vessel dispersant application system, such as its swath, on-board dispersant storage capacity, and pump rates, and you describe the circumstances of its operation, such as cascading information and transit distance to the oil spill area and speed (*see Section Three, EDSP Calculator Inputs*). The Calculator then estimates the amount of dispersants that can be applied and the amount of oil that the system could treat during each of the first two days after a spill occurs (see *Section Four, EDSP Calculator Outputs*). These estimates of the oil treated are termed the "Estimated Dispersant System Potential" (EDSP) for the given dispersant application system configuration. The use of the calculator tool in both spill modes is further demonstrated and explained in *Section Five, Using the EDSP Calculator*.

The EDSP Calculator was developed with the intent of reinforcing incentives for creating and acquiring more effective dispersant application systems, and in addition, evaluating the potential of a dispersant application system to meet various regulatory planning requirements. Experimentation with the Calculator is helpful in understanding the effects of different configurations on a system's treatment potential, and may provide incentives for developing more effective dispersant application systems.

1b. Dispersant Application System Components

The EDSP Calculator was designed as a planning tool for estimating the potential of aircraft and vessel dispersant application systems. The major components of a dispersant application system include a dispersant storage tank, a pump system to move liquid dispersant from the storage tank to the distribution system, a power source, spray arms, and nozzles. Figure 1 below identifies the major components of a dispersant application system:

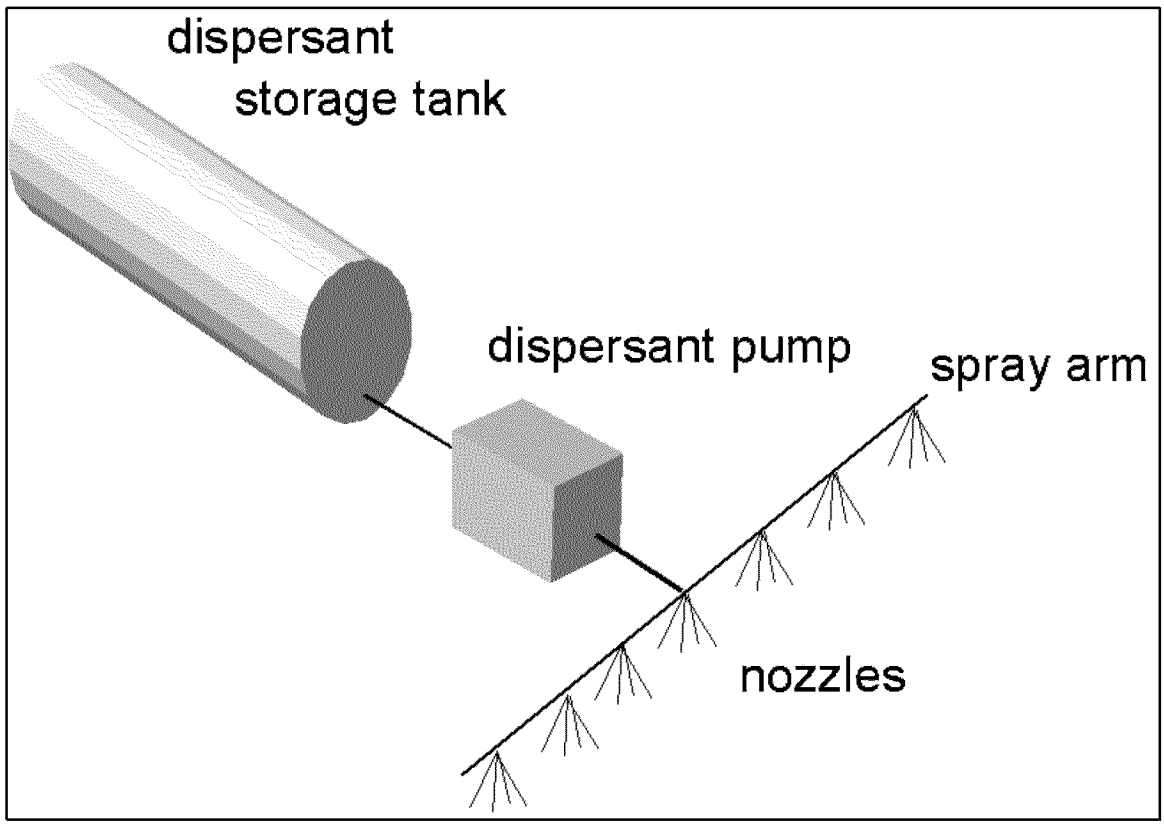


Figure 1. Dispersant Application System Components.

Below are short descriptions of the various components of a dispersant application system:

Delivery platform aircraft or vessel –This is the supporting platform for all the components of a dispersant application system including the following: dispersant tanks, dispersant pumps, delivery piping and manifolds, spray arms, and nozzles.

Onboard dispersant storage – Includes tankage built into or carried on the delivery platform to hold the dispersant. A payload is defined as a complete fill of the tank.

Pump – The pump transfers the dispersant from the tank through piping and manifolds to nozzles on the spray arms. The pump rate and nozzle configurations are adjusted to

deliver the dispersant at the desired dosage. Check the platform spray chart to insure that it can deliver the desired dosage and calculated pump rate.

Spotter aircraft - While not shown in Figure 1, spotter aircraft can be considered to be part of the "system" for dispersant application. Aerial delivery platforms necessarily fly low and at these altitudes oil slicks are difficult to see. The spotter aircraft assists in spill detection, tracking, alignment of the dispersant platform with the oil slick, and application-determination of "spray on" and "spray off."

The following are photographs of some dispersant application platforms:

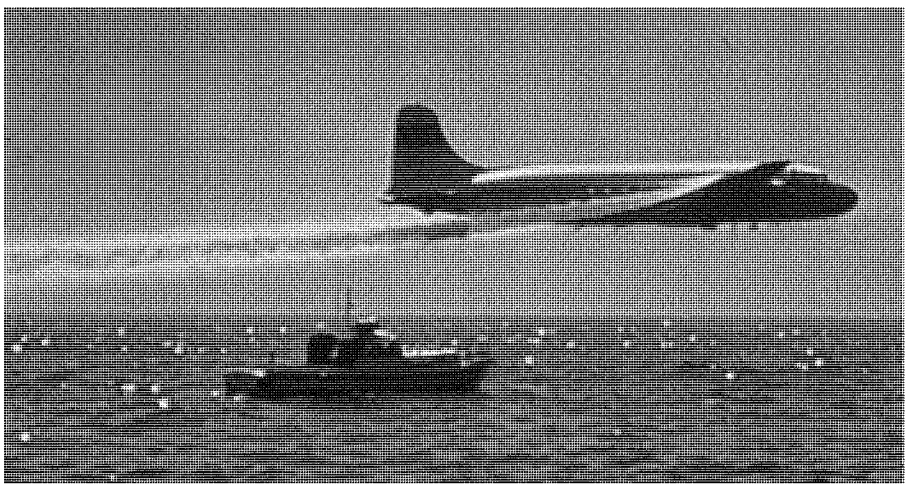


Figure 2. DC-4

Figure 3. C-130 Hercules

Figure 4. Helicopter with Dispersant Spray Bucket

Figure 5. Large Dispersant Vessel

Figure 6. Small Dispersant Vessel

2. Assumptions and Limitations as a Planning Tool

The EDSP Calculator was developed to provide a distribution-rate, system-based estimate of daily dispersant application. The calculator accommodates a broad range of dispersant system configurations and addresses response activities including the accessing, spraying, and treatment of oil. The Calculator also accounts for the transiting of a dispersant application system to and from a staging area for reloading of dispersant and fuel. The goal was to provide a computer tool that could facilitate the calculation of a "Planning Standard", not a "Performance Measure".

The ERSP, EBSP, and EDSP Calculators are intended as planning tools for estimating the potential of different oil spill response systems to mitigate (recover, burn or disperse) discharged oil relative to one another. These planning tools are NOT intended to be used as models for calculating system performance during an actual oil spill, which is affected by many factors such as the distribution of oil on the water surface, oil weathering, and other ambient on scene conditions which are not included in these Calculators.

The following is a list of assumptions and limitations inherent to the design of the EDSP Calculator, that are readily acknowledged as conditions accepted in order to keep the Calculator a simple and easy to use planning tool:

2a. General Assumptions

- ***Ambient Conditions:*** Estimates made by the Calculator assume that conditions are generally conducive to effective dispersant application operations. Its output is designed to serve as a guide for planning the deployment of dispersant application systems and estimating their delivery potential.
- ***System Downtime:*** No downtime due to maintenance or repair is considered.
- ***Oil and dispersant types:*** The Calculator does not differentiate outputs based on the type of oil or product being treated nor the type of dispersant being used. The Calculator assumes that the oil in question for the scenario, spray system, and dispersant being used is dispersible commensurate with expected efficiency and effectiveness ranges.
- ***Success of Dispersion:*** The EDSP Calculator does not provide information relating to the ultimate success of chemical dispersion of oil, but rather only the estimated potential volume of oil treated.
- ***Oil Thickness on the Water:*** For the purposes of planning and comparing spray systems, the Calculator does not assume any specific oil slick thickness will be encountered, however, an oil slick thickness of 0.004 inches (0.094 mm) is equivalent to the thickness of oil that would be treated by spraying dispersant at a dosage rate of 5 gallons per acre with a dispersant to oil ratio of 1:20.
- ***Dosage:*** The Dispersant Aircraft Capability Forms (see Figure 24) do not contain explicit dosage capability values or ranges. Every aircraft platform in the EDSP database can deliver the regulatory dosage of 5 gallons of dispersant for each acre sprayed. Other dosage values used in the EDSP may not be actually achievable and should be verified with the platform operator before use.
- ***The Use of Default and Alternate Values:*** The Calculator, and its default values for many of the key inputs, is intended to be "conservative" in nature. In some cases, a dispersant application system's tested performance will be able to exceed the default values for specific inputs. Plan holders or OSROs may submit requests for the use of alternate values in the Calculator to the appropriate regulatory agency. The regulatory agencies will consider each request based on the merits of the documentation provided. Operators should expect that the use of approved alternate values may be subject to validation efforts by regulators with plan holders and OSROs being required to satisfactorily demonstrate the practical viability of these values during equipment verification visits and PREP equipment deployment exercises.
- ***Incident Briefing:*** All dispersant application platforms will hold an incident briefing for 45 minutes prior to departing the staging site to commence the first spray sortie. Normally this briefing will only occur in Operating Period1 (OP1), however, for a platform that cascades into the scenario and arrives for its first sortie in a subsequent OP that initial briefing will occur in the OP that contains the first operational spraying sortie.
- ***Check-in Time:*** Check-in time of 10 minutes with spotter aircraft occurs at the beginning of each sortie for aircraft platforms.

- ***Side Overlap and Gaps:*** Spray passes will not be precise in the real world. There will be side overlap and gaps in the spray pattern between passes in some cases, as application platforms will strive to only spray areas containing oil, which usually will not be continuous and uniform in nature on the water's surface. No side overlap or gaps are assumed in the Calculator.
- ***Resupply On Scene:*** When "Resupply On Scene" is set to "Yes" for vessel-based application systems, dispersants will be resupplied onsite at the spill location to the platform and application system, either when the payload of dispersant is completely applied during spraying operations, or when the operating period is completed, whichever occurs first.
- ***Resupply at the Staging Site:*** When "Resupply On Scene" is set to "No" (which is the default setting) for vessel-based spray application systems, the vessel will resupply its dispersant payload at the staging site. This will require the spraying vessel to transit between the spill site and the staging site, similar to aircraft operations, whenever more dispersant payload is needed. When "Resupply On Scene" is set to "No", if a vessel has less than one half of its dispersant payload left onboard when the OP ends, the platform will immediately transit back to the staging area to resupply. However, if the vessel has more than one half of its payload remaining onboard at the end of the OP, the platform will stay at the spill site between operating periods and will commence spraying the remainder of the payload at the beginning of the next operating period.
- ***Loading in between Operating Periods:*** At the end of each Operating Period it is assumed that aircraft-based spray systems will reload dispersant and fuel before the next Operating Period begins.

2b. Additional EDAC assumptions and scenario:

- "Platform" inputs are reset and locked to default values
- The Dispersant-to-Oil Ratio (DOR) is 1:20.
- Five gallons of dispersant will be applied to each acre (46.8 liters/hectare).
- The pass length for the oil slick to be treated is 4 nautical miles (7.4 km) long in the direction of spraying.
- Aircraft platforms have 12 hours of time available for Operating Period 2 dispersant operations, not including the last transit back to the primary dispersant staging site.
- Cascading of aircraft and vessels from out-of-area requires entries for the platform/dispersant mobilization time, cascade distance, and platform dispersant payload status (loaded or empty). Cascading reduces the Operating Period resulting in the use of Adjusted Operating Period 1 for EDSP calculations.
- Aircraft platforms will be allowed to apply dispersants in both directions.
- Spotter aircraft are assumed on-scene waiting for dispersant spray platforms.

- Aircraft platforms will be allowed to refuel and load dispersants at the same time except for USAF Reserve C-130 H.
- Transit distance for vessel response plans is measured from the primary dispersant staging site (or closest dispersant staging site for plans with multiple staging sites) to the furthest spill site 50 NM off shore within a Captain of the Port (COTP) zone.
- Transit distance for facility response plans is measured from the primary dispersant staging site to the facility.
- Weather: Visibility greater than 3 statute miles (4.83 kilometers)* Ceiling greater than 1000 feet (304.8 meters)* No wind (0 mph) No precipitation Mild temperatures 40-80 degrees F (4.4 – 26.7 degrees Celsius) *Visible Flight Rule (VFR) conditions are required at the spill site for aircraft to apply dispersants and are assumed for all flight operations, including cascading from home base to staging airports and for transit to and from the spill site. VFR flight requires 30 minutes of fuel reserve on the aircraft.

3. EDSP Calculator Inputs

This section provides a general description, and in many cases, amplifying guidance, for each of the input fields and the variables that a user must enter into the Calculator. The graphic below is an illustration of the input screen that a user will see and use to enter their dispersant system information into the Calculator.

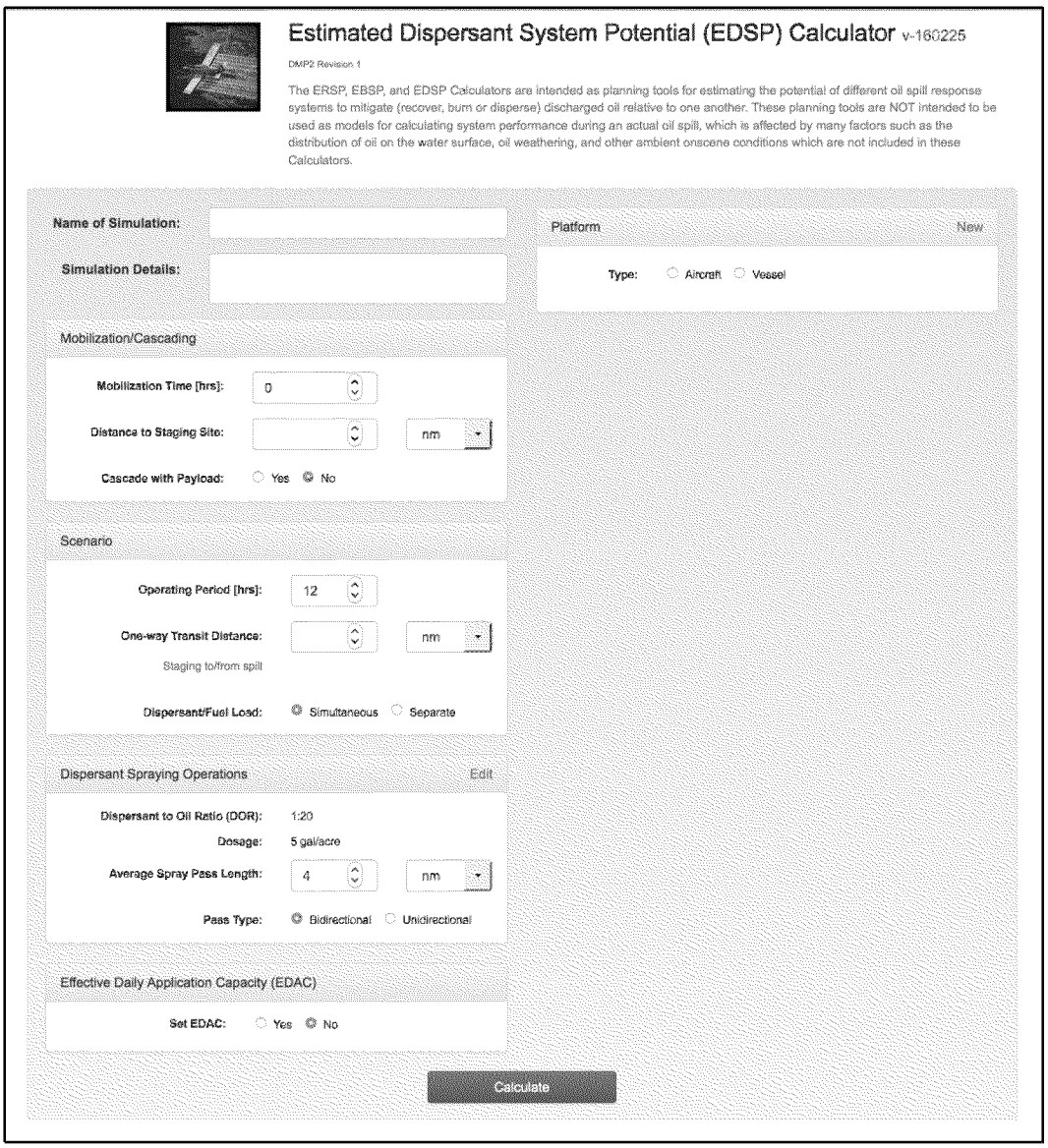


Figure 7. EDSP Calculator Input Screen

- Name of Simulation: Entry field for the name or other form of identifier for the dispersant application system (up to 48 characters).

- Simulation Details: Enter configuration details including the type of platform, skimmer, pump, and boom being used and other key information to identify this simulation.

The EDSP Calculator inputs have been grouped into five overall categories: Mobilization/Cascading, Scenario, Dispersant Spraying Operations, Effective Daily Application Capacity (EDAC), and Platforms

3a. Mobilization/Cascading Inputs

Figure 8. Mobilization/Cascading Input Screen

- ***Mobilization Time:*** As resources are ordered at the onset of an oil spill response, each individual dispersant platform and/or dispersant stockpile will need a time period for activation and mobilization. Enter the time necessary for activation and mobilization to include all activity necessary to get the resource moving. Use the greater of the platform mobilization time or the dispersant stockpile mobilization time for use in the EDSP unless the platform will be cascading with payload If the platform is cascading with payload, use the mobilization time for the spray platform. Mobilization Time should be based on:

    - Dispersant stockpiles must be available onsite or already loaded onboard the spray platform before it can depart for its first dispersant spraying sortie. For situations where a platform is cascading to a staging site without payload, mobilization times for dispersant stockpiles should be calculated based on the distance from the stockpile location to the selected dispersant operations staging site using applicable transit routes and speeds. For situations where a spray platform is departing from its home base to conduct sortie operations, if dispersants are not already

onsite, stockpile mobilization times must be calculated from the dispersant storage location to the spray platform's home base. The default transit speed for moving dispersant stockpiles to staging sites is 35 mph over land or 100 kts when moving stockpiles by air.

- Dispersant oil spill removal organizations should use the time specified in their contract for activation and mobilization of dispersant spraying platforms, or the time listed in other documentation that has been deemed acceptable by the appropriate regulatory agency.
- For situations where activation and mobilization times for spray platforms are not specified in a contract, the following table should be used for estimating activation and mobilization time:

Platform Activation and Mobilization Time (In Hours)		
Spraying Platform Status	Response Personnel Availability	
	24 hour On Site	Recall
Owned/Dedicated	1	2
Contracted/Dedicated	1.5	2.5
Owned/Non-dedicated	2.5	3.5
Contracted/Non-dedicated	3	4
Dedicated means used solely for the purposes of spill response and dispersant spraying operations. *Contracted/Owned refers to the relationship between the organization providing the dispersant spraying service and the spraying platform.* *Response availability applies to the full complement of personnel that would be required to mobilize and operate the spray platform.*		

Figure 9. Platform Activation and Mobilization Time

- ***Distance to Staging Site:*** If cascading from out of area or from a home base to a dispersant operations staging site (the staging site is where the aircraft will return for loading dispersants and fuel between sorties), the user enters a cascade distance, and specifies the distance units. The cascading calculation for all platforms includes a 45 minute briefing period (platforms are assumed to be loading fuel and dispersants as necessary during the 45 minute briefing period). Enter a numerical value here for the distance if the platform will be cascading from out-of-area to the incident staging site. The default distance unit is nm (nautical mile). Kilometers (km) or statute miles (mi) can be selected via a drop down list.

- ***Transit with Payload:*** If cascading from out of area or from a home base to a staging site, the user enters a cascade distance and specifies if cascading occurs with payload or with no payload. Most aerial platforms have a different Cascade Transit Speed and Maximum Range depending on whether the platform is cascading with or without a dispersant payload. With a selection of Yes, the Calculator will use the range and speed values in the database for cascading with a payload.

3b. Scenario Inputs

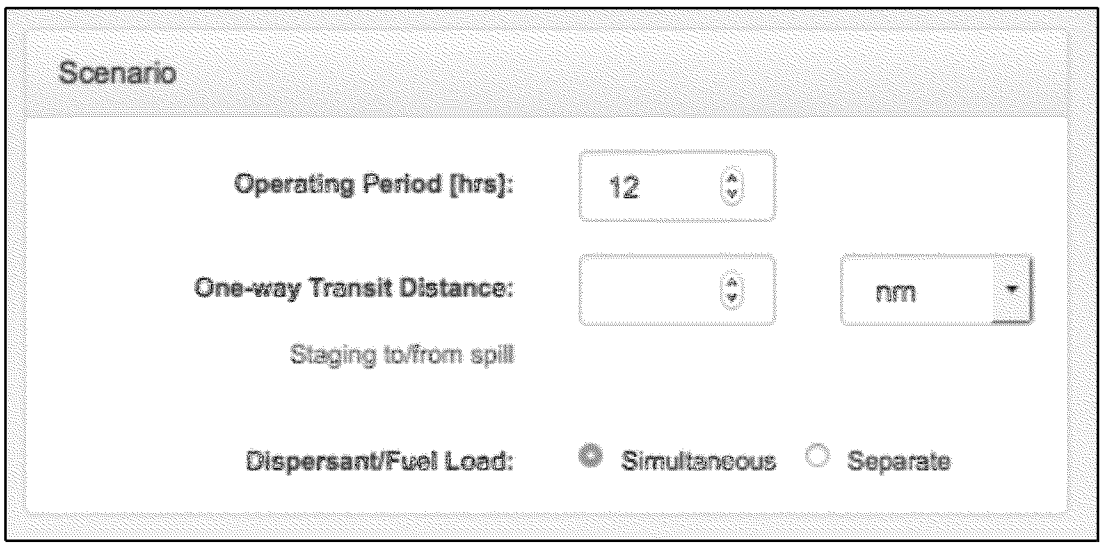


Figure 10. Scenario Input Screen

- ***Operating Period [hrs]:*** The Operating Period, which should not be confused with an operational period for an Incident Action Plan, is the length of time (in hours) each day where conditions allow a dispersant application system to conduct spraying operations. The first Operating Period begins coincident with the starting time for the mobilization of the spraying platform and lasts for twelve hours (unless an alternate OP length is approved for the scenario being evaluated). The second OP will start at the twenty-four mark and ends at the thirty-six hour mark (see diagram below).

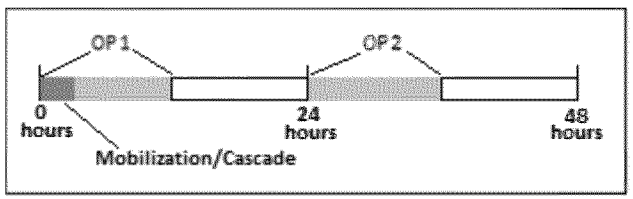


Figure 11. Operating Period

The default Operating Period (OP) is 12 hours. The OP will be reduced by Mobilization Time and any necessary cascade transit time. Alternative values for the length of the Operating Period may be requested, for example, when a plan is for an area and season that has extended hours of daylight. Requests for an extension of the Operating Period would be evaluated on a case-by-case basis by the appropriate regulatory agency.

- ***One-way Transit Distance (Staging area to/from spill):*** The staging area is the location where platforms will reload fuel (as necessary) and dispersant at the conclusion of each sortie. The one-way transit distance is the distance from the staging area to the spill location. The transit distance used for EDAC calculations for vessel response plan is measured from the primary dispersant staging site to the spill location furthest away within the boundaries of the Captain of the Port (COTP) zone that could occur 50 nm offshore. For more detailed information on estimating one way transit distances for use in EDAC calculations, see Section 3d of this user manual, "EDAC Inputs".
- ***Dispersant/Fuel Load:*** Dispersants and fuel can be loaded separately (sequentially) or at the same time (simultaneous). If Simultaneous is selected, the greater of the fuel load time and the dispersant load time will be used by the Calculator.

3c. Dispersant Spraying Operations Inputs

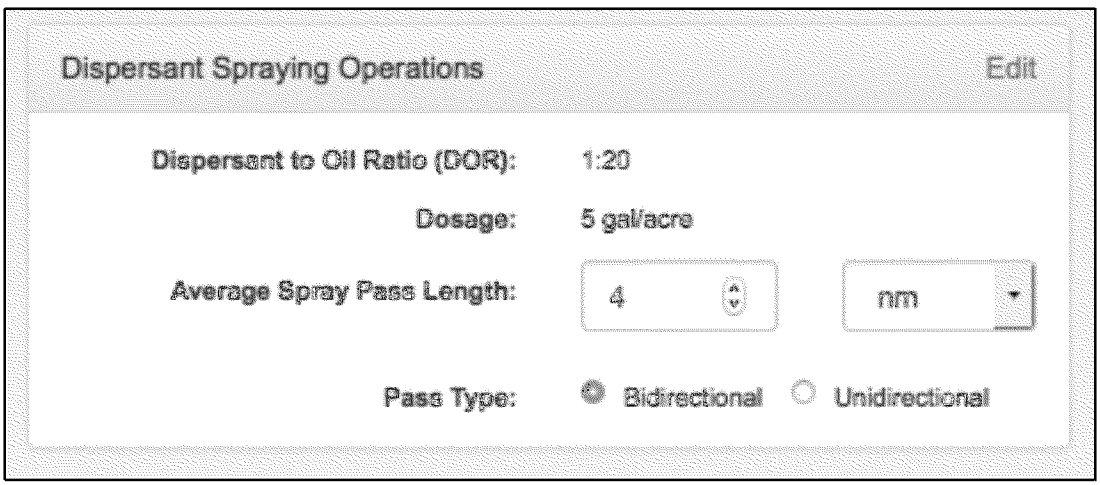


Figure 12. Dispersant Spraying Operations Input Screen

- ***Dispersant to Oil Ratio (DOR):*** The volume of oil that one volume of dispersant will treat is expressed by the DOR. For example, a DOR of 1:20 means that one volume of dispersant will potentially treat 20 volumes of oil. The DOR can be changed by clicking on the Edit button located at the upper right of the Dispersant Spraying Operations Group. Currently the default DOR is 1:20.

Note: In the future, BSEE-regulated users may be asked to enter the DOR based upon the information available in the order of precedence:

- The minimum DOR that meets EPA-effectiveness thresholds, based upon an oil characterization and dispersibility test of the oil handled at your facility and the dispersant that you have stockpiled and ensured available in your OSRP.
- If oil characterization and dispersibility test information are not available, enter the manufacturer recommended DOR (if given as specific single value); if the manufacturer does not provide a specific DOR, but instead provides a recommended overall range, then use 1:20. This information may be found in the National Contingency Plan Product Schedule maintained by EPA at: http://www.epa.gov/sites/production/files/2013-08/documents/schedule.pdf.

Note: The following information is currently provided by manufacturers regarding their dispersants and recommended DORs:
Corexit EC9500A: A dispersant to oil ratio of **1:50 to 1:10** is recommended.
Finasol OSR 52: A dispersant to oil ration of **1:5 to 1:50** is recommended, according to the nature of the oil, its state of weathering, temperature, thickness of the slick, and other circumstances of the spill.
Accell Clean DWD: A dispersant to oil ratio of **1:10** is recommended. The rate varies depending on the type of oil, degree of weathering, temperature, and thickness of slick.
Dispersit SPC 1000: A dispersant to oil ratio ranging from **1:50 to 1:10** is recommended. These rates will be dependent on the type of oil, degree of weathering, temperature and extent of oil slick.

- ***Dosage:*** Dosage is the volume of dispersant that is applied to the specified area. The default is 5 gallons of dispersant applied to every acre sprayed.

  The default DOR and dosage values can be changed using the Edit button (see Figure 11) at the upper right hand corner of the Dispersant Spraying Operations Input Screen. When the Edit button is selected, the Dosage/DOR pop-up edit screen is displayed as illustrated below in Figure 13.

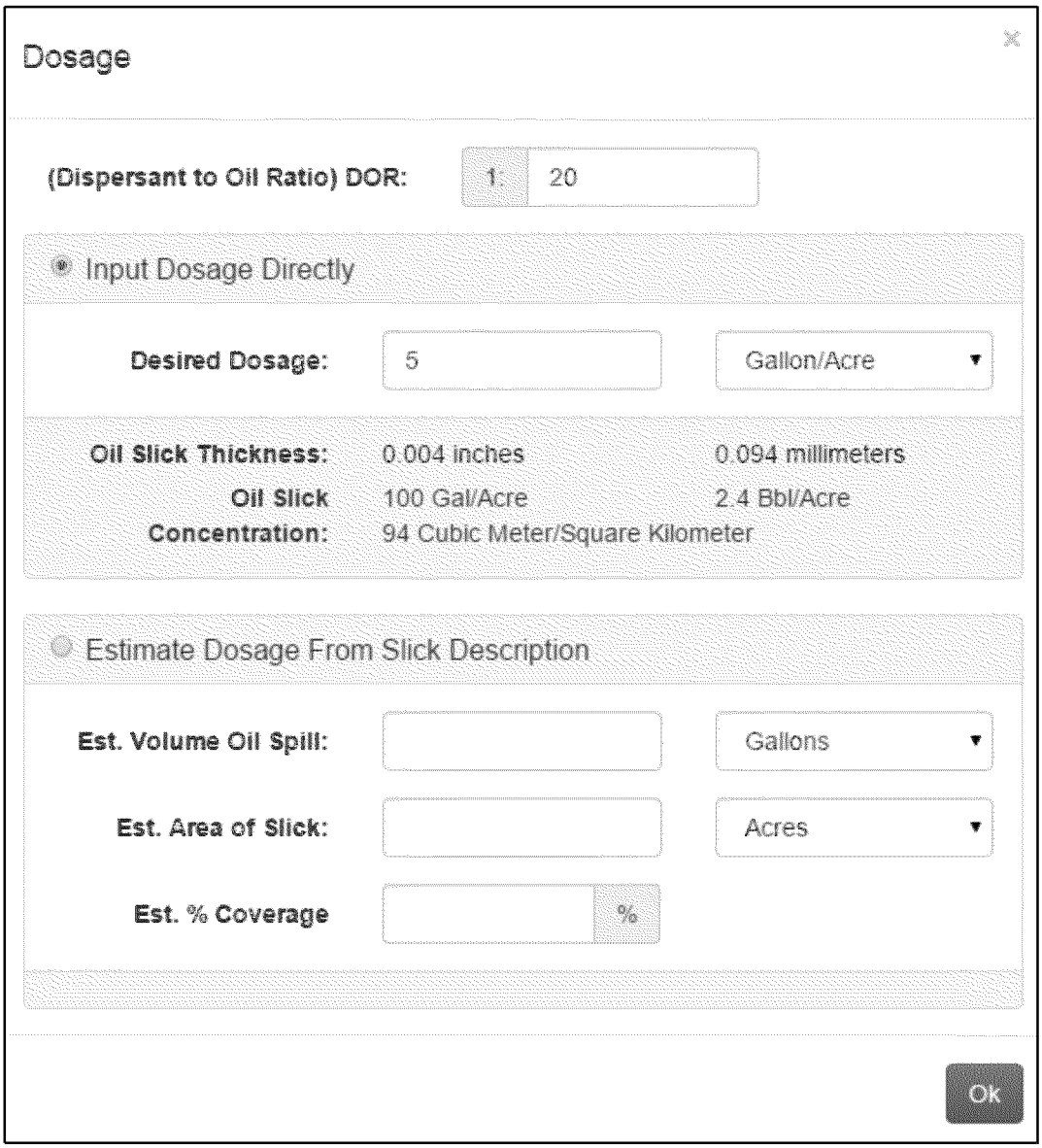


Figure 13. Dosage/DOR Edit Popup Screen

- ***Average Spray Pass Length:*** _This is the average length of the oil slick that is parallel to the spray path of a platform. The default Average Spray Pass Length is 4 nautical miles.
- ***Pass Type:*** The Pass Type can be Bidirectional or Unidirectional. Bidirectional means the aircraft platform sprays in both directions over the oil slick. Unidirectional is sometimes called "Race Track" mode in which an aircraft platform sprays in one direction only (usually upwind) over the oil slick. On the return leg, the aircraft transits over the oil slick without spraying. Figure 14 illustrates aircraft bidirectional spraying and Figure 15 illustrates aircraft unidirectional spraying

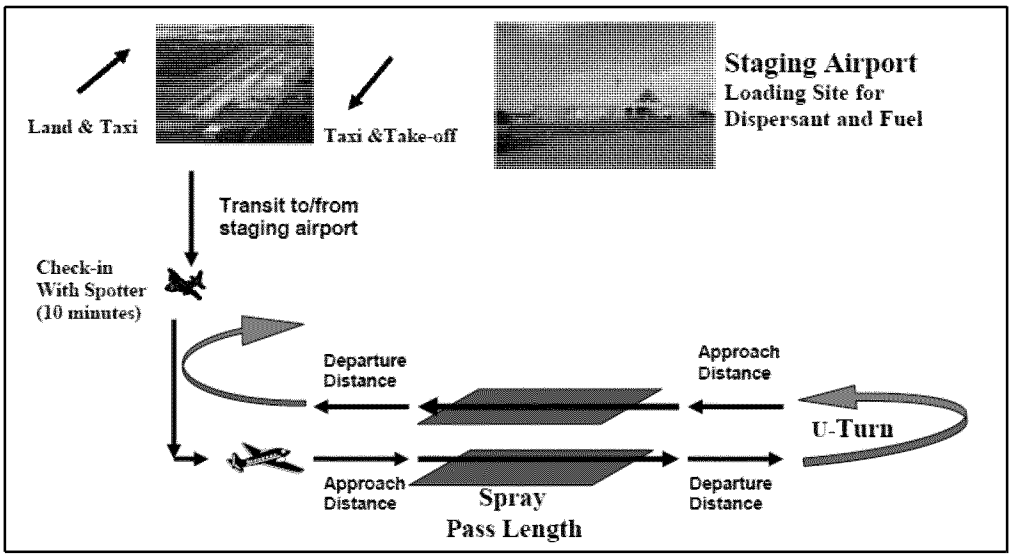


Figure 14. Bidirectional Spraying

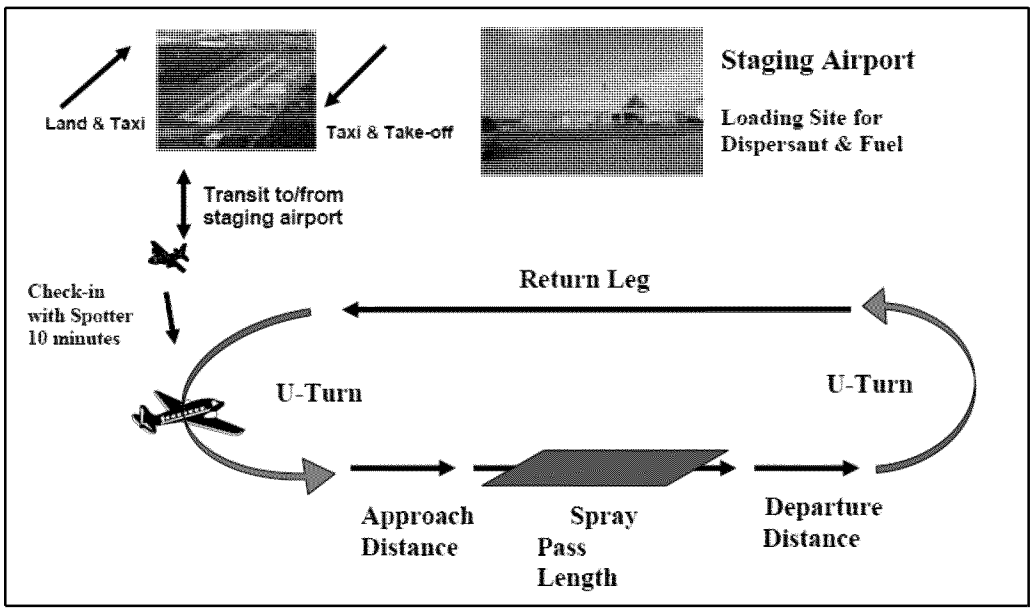


Figure 15. Unidirectional Spraying

3d. Effective Daily Application Capacity (EDAC) Input

The requirements for certain Coast Guard-regulated oil spill response plans have regulatory thresholds for the ability to apply dispersants during the initial days of a spill. These regulatory thresholds are called "Effective Daily Application Capacity" (EDAC). This EDSP Calculator was designed to provide both EDSP and EDAC outputs when necessary through the use of an EDAC radio toggle button. Selecting the "Yes" radio button ensures certain regulatory assumptions about inputs are maintained and provides computed EDAC values as outputs.

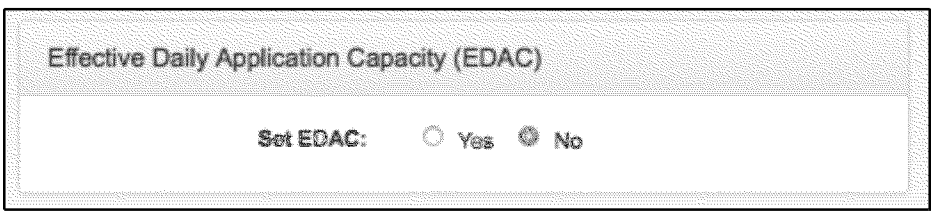


Figure 16. EDAC Selection Screen

While the default setting for the EDAC button is "No", the EDSP Calculator will adjust certain calculator inputs for calculating EDAC when the EDAC "Yes" button is selected. Selecting "Yes" will reset and lock the following input values necessary for calculating EDAC:

Scenario Inputs

- Operating Period will be reset and locked at 12 hours.
- Dispersant/Fuel Loading will be set to simultaneous.

Spraying Inputs

- Average Pass Length will be reset and locked at 4 nautical miles.
- Pass Type will be set to Bidirectional.
- DOR will be reset and locked at 1:20.
- Dosage will be reset and locked at 5 gallons per acre.

Platform Inputs

- Default values for the selected dispersant application platform will be reloaded and locked. No edits to these values will be permitted, and the cursor will lock when placed over any of the platform input fields.

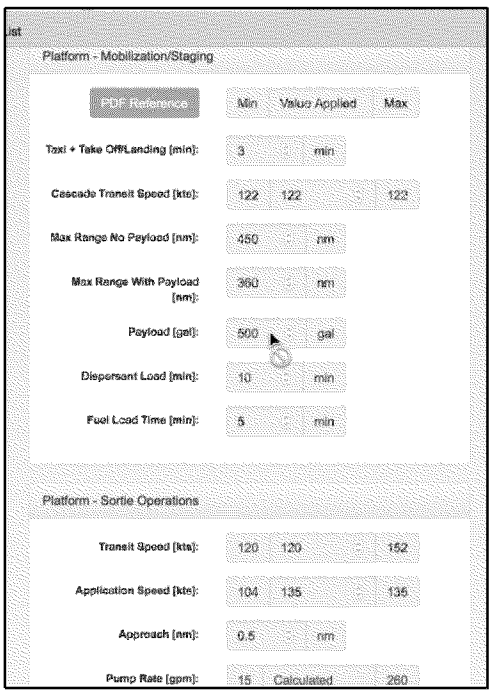


Figure 17. Locked EDAC Values

Calculating One Way Transit Distance for EDAC

When calculating one way transit distances for Facility Response Plans (FRPs), the distance from the staging site to the facility should be used.

When calculating EDAC for Vessel Response Plans (VRPs), dispersant spraying capabilities should cover the entire Captain of the Port (COTP) Zone out to a distance 50 nautical miles from shore. For a spraying platform that will only use a single primary staging site, the one way transit distance will be the longest distance possible that can be measured from the staging site to the farthest point in the COTP zone that is 50 miles offshore. Figure 17 below illustrates how to calculate the one way transit distance in such cases:

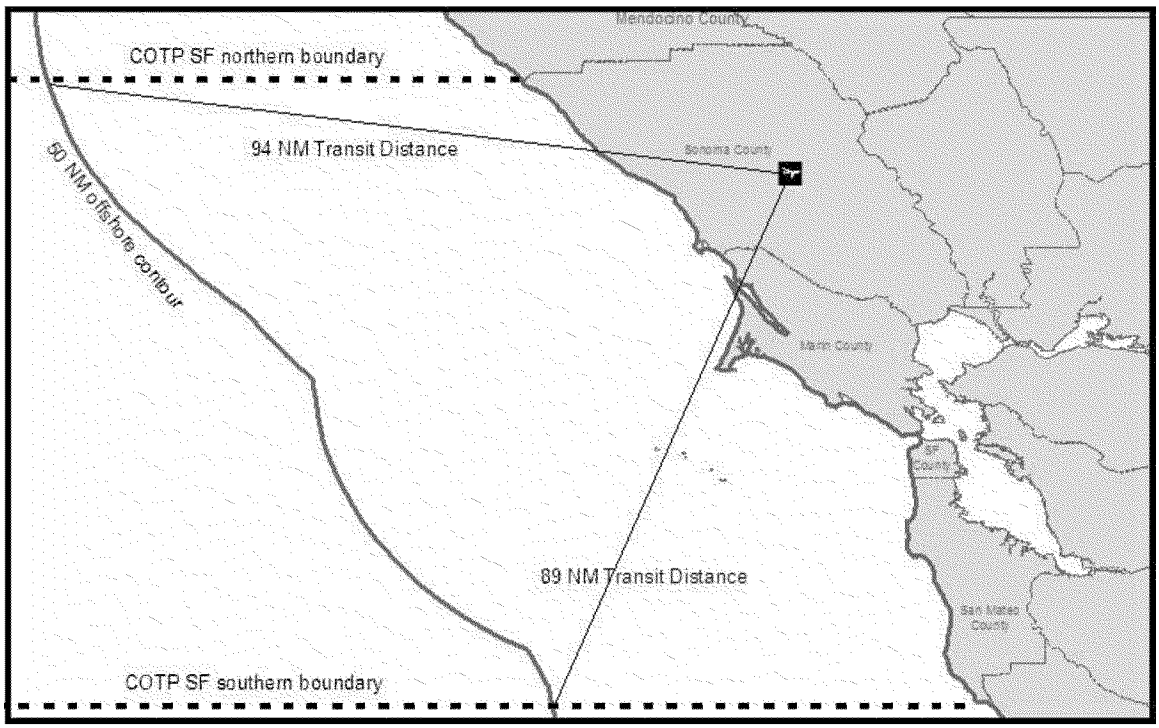


Figure 18. Example of EDAC Transit Distance Using a Single Staging Site.

The illustration above demonstrates how to determine the EDAC one-way transit distance (when calculating dispersant spraying coverage for a VRP) using the example of COTP San Francisco and the Sonoma County Airport as a single primary dispersant staging point. The distance from the staging point to the furthest extent of the 50 NM off shore contour is measured as 94 nautical miles.

Multiple dispersant staging sites may be specified in a VRP. The COTP zone could be subdivided into areas that would be covered by each staging site. Using the COTP San Francisco as an example, the Mendocino County Airport and the Half Moon Bay Airport could be used as multiple staging sites. If the Mendocino County Airport was the only staging site specified in the plan, the EDAC Transit Distance would be 124 NM. The EDAC Transit Distance for Half Moon Bay would be 131 NM if it were the only staging site specified in the plan. However, if both airports were specified as staging sites in the plan, the COTP zone could be divided into two areas by a line equidistant from the two airports. The diagram shows the San Francisco COTP zone with a dispersant staging site at the Mendocino County Airport for the northern area of the COTP zone with a EDAC Transit Distance of 85 nautical miles and a dispersant staging site at the Half Moon Bay Airport for the southern area of the COTP zone with a EDAC Transit Distance of 72 nautical miles.

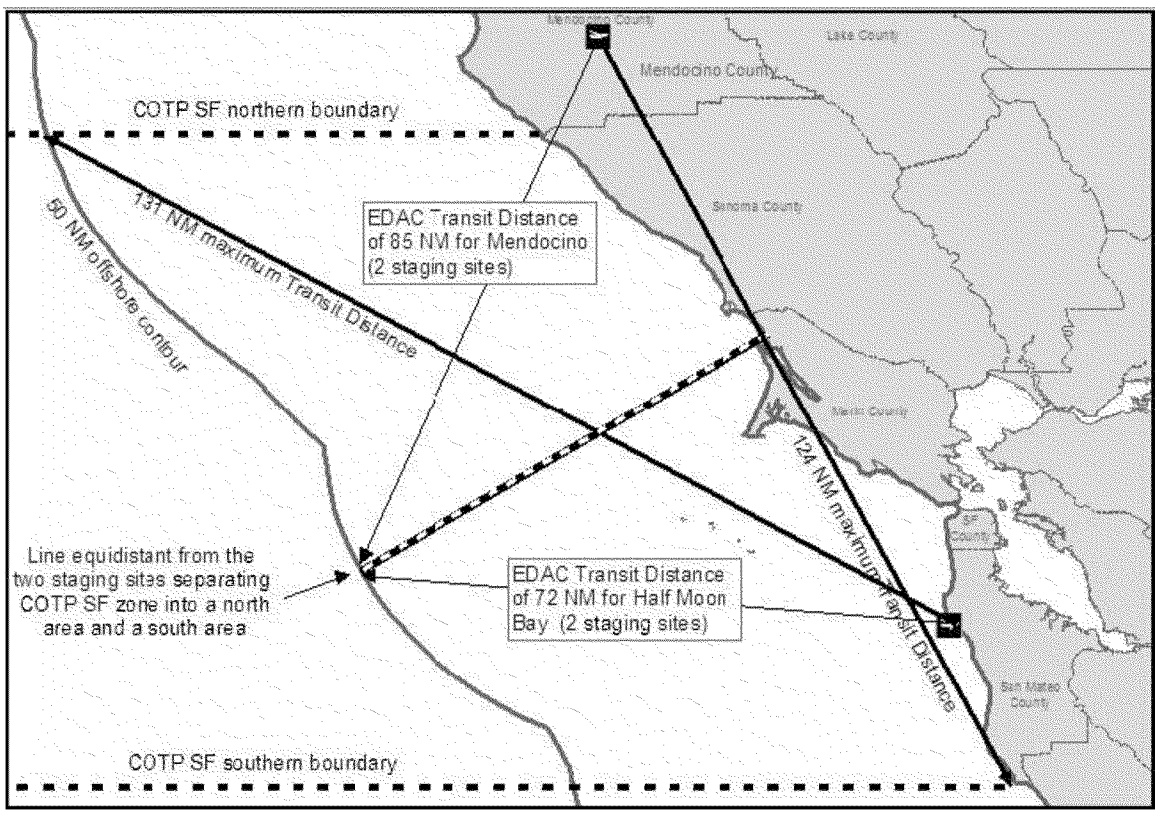


Figure 19. Example of EDAC Transit Distance for Multiple Staging Sites.

3e. Platform Inputs

The EDSP Calculator contains an internal database of spray platforms and their operational characteristics, such as transit speeds, dispersant payloads, etc. These platforms have been tested and vetted regarding their operational characteristics, and the listed default values for these platform operational inputs are already accepted and approved for use by BSEE and the USCG in the EDSP Calculator.

The EDSP Calculator also allows users to either 1) create new platforms based on the characteristics of the spray platform and system not currently listed in the database, or change any of the existing default values listed for a platform in the database in order to more accurately represent the capabilities of a platform and spray system that has been modified. When submitting any EDSP/EDAC calculations to BSEE or the USCG, users should clearly identify any platform input values that have been modified from the existing default values, and to show documentation demonstrating the validity of the alternate values that have been proposed for use in the calculator. Users must obtain approval from the appropriate regulatory agencies for the use of new platforms or any alternative input values to be used in conjunction with any existing platforms in the EDSP Calculator.

Platform Type (Aircraft or Vessel):

In the EDSP, you must choose between an aircraft or vessel-based spray system as illustrated in the figure below. Once you select a type, such as “Aircraft” shown below, a drop down menu appears that will allow you to select a specific type of platform. This selection will determine the format of the inputs the Calculator will display, as the input fields for aircraft and vessels are different.

Figure 20. Platform Input Screen.

Aircraft Selection

To select an aircraft platform, click on the “Select Aircraft” drop down menu, and then select a specific aircraft from the Calculator’s database of existing spraying platforms.

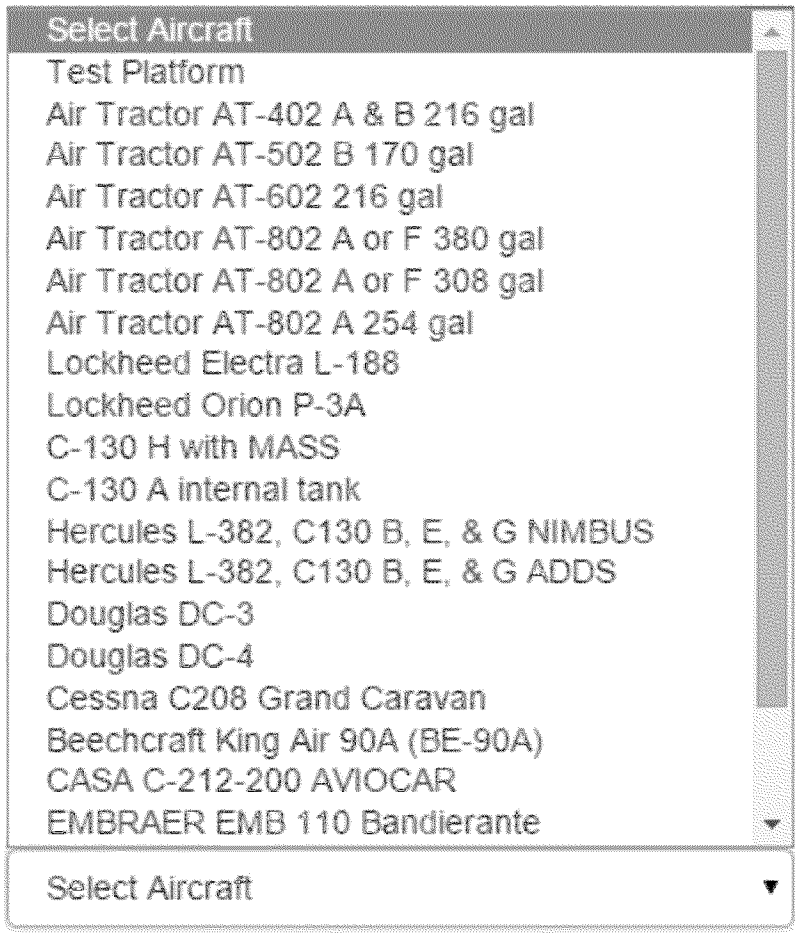


Figure 21. Existing Aircraft Platforms.

Aircraft Platform Input Screen

Once an aircraft platform has been selected from the platform dropdown menu, the default calculator input values (and associated ranges) are displayed in two groups: *mobilization/staging* and *sortie operations*.

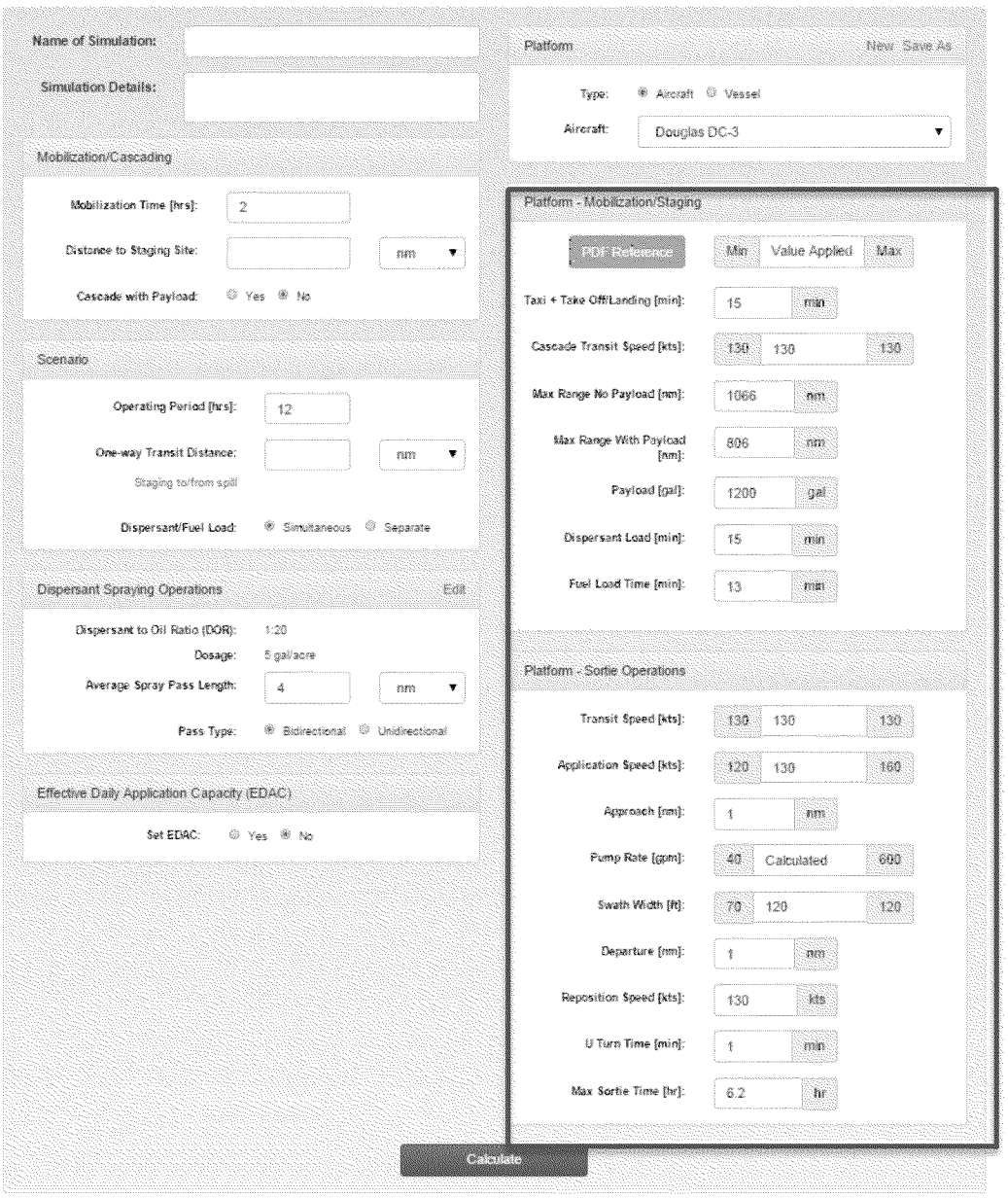


Figure 22. Aircraft Platform Input Screen.

Aircraft Mobilization/Staging Input Fields:

Figure 23. Aircraft Platform – Mobilization/Staging Input Group

When selecting a specific aircraft platform, default input values (or ranges of values) are provided for the following platform parameters:

- Taxi + Take Off/Landing [min]: The time in minutes for the selected aircraft platform to taxi prior to take off and to taxi after landing.
- Cascade Transit Speed [kts]: The speed at which the aircraft transits while cascading from out of area. For most platforms, this speed will vary depending on whether the cascade is with or without a dispersant payload. Three values are displayed; the minimum Cascade Transit Speed, The value used for calculations, and the maximum Cascade Transit Speed.
- Max Range No Payload [nm]: This is the maximum range this platform has with maximum fuel load and no dispersant payload.

- Max Range With Payload [nm]: The maximum range for this platform with full dispersant payload and maximum takeoff weight.
- Payload [gal]: The maximum volume of dispersant the selected platform can carry.
- Dispersant Load [min]: The time necessary for a complete dispersant load.
- Fuel Load [min]: The time necessary to load fuel onboard the platform.

Aircraft Sortie Operations Input Fields:

Figure 24. Aircraft Platform - Sortie Operations Group.

- Transit Speed [kts]: Three values are displayed; the minimum Transit Speed, the value used for calculations, and the maximum Transit Speed.
- Application Speed [kts]: Application speeds (averaged between passes) are assumed to be over-the-ground. It is acknowledged that all aircraft have a range

of acceptable air speeds for safe operations; however, when the effects of wind are considered the resulting ground speeds may vary considerably. Wind speed variation is not taken into account in the EDSP Calculator.

- Approach [nm]: The Approach is the distance over which the aircraft aligns on course for the next spray pass and descends to spray altitude at Reposition Speed prior to commencing dispersant spraying. See Figures 12 and 13, Race Track and Bi-directional Application Diagrams.
- Pump Rate [gpm]: The calculated Pump Rate necessary to deliver the required dosage. The EDSP checks to insure that the calculated pump rate is between the maximum and minimum pump rates and assumes that the selected platform can deliver the calculated pump rate. Note: check the spray chart for the selected aircraft to insure that it can deliver necessary pump rate and dosage.
- Swath Width [ft]: The length measured perpendicular to the dispersant application track between the points at which delivery on the oil slick drops below 90% of the designed dosage (ASTM Standard F 1413-07 (2013)).
- Departure [nm]: The distance the aircraft continues on course and at Reposition Speed before commencing a U Turn. See Race Track and Bi-directional Application Diagrams.
- Reposition Speed [kts]: The speed at which the aircraft performs Approaches, Departures, and Return Legs.
- U Turn Time [min]: The time necessary for the aircraft to make a 180 degree turn after a Departure and before the Approach for the next spray pass.
- Max Sortie Time [hr]: The maximum flight time the aircraft can achieve assuming no wind, VFR fuel reserve, and with a full dispersant load.

Aircraft PDF References

Clicking on the blue PDF Reference button will display the Dispersant Aircraft Capability Forms for the aircraft platform that has been selected. There forms contain detailed information regarding the aircraft performance parameters as well as information about the sources of the data. The data in the form column labeled "U.S. Regulatory Calculation Values" were used to populate the DMP2/EDSP Calculator database for the aircraft platform default input values. These default values are used by the Calculator's algorithms to calculate EDSP and EDAC.

Figure 25. PDF Reference Button

The following example (Figure 26A) shows the various data fields and forms within the Capability Form for the Beechcraft King Air (BE-90A) .

DISPERSANT AIRCRAFT CAPABILITY FORM

PLATFORM

BEECHCRAFT
KING AIR 90A (BE-90A)

Operator: Dynamic Aviation, Inc.
OSRO: Marine Spill Response Corp.

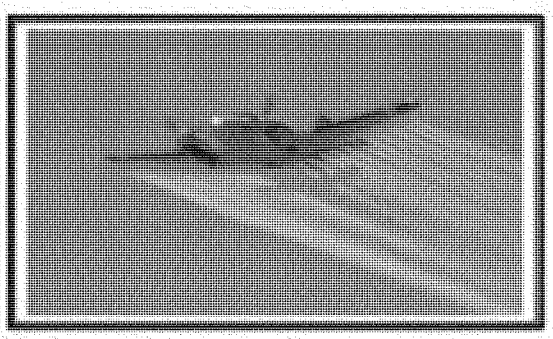


Photo compliments of the Marine Spill Response Corporation

DATA SOURCE LEGEND

1. (Black): Indicates the data are based on documented field trials or is a fixed design value
2. (Blue): Indicates the data are based on limited field observations or operator's stated practice or stated value (little or no documentation)
3. (Red): Indicates the data are based on reasonable calculations or performance of comparable systems

		Unit	U.S. Regulatory Calculation Values	Data Source 1-2-3	Range	Reference(s)
	AIRCRAFT PARAMETERS					
1	Swath Width	feet	75	1	60-85	Dynamic Aviation Group, Inc. spray test May 06
	a. Application (gallons per acre)	gpa	5	1	1-10	Dynamic Aviation Group, Inc. spray test May 06
	b. Altitude	feet	75	1	50-100	Dynamic Aviation Group, Inc. spray test May 06
	c. Application Speed	knots	150	1	130-175	Dynamic Aviation Group, Inc. spray test May 06
	d. Pump Rate (gallons per minute)	gpm	85	1	20-180	Dynamic Aviation Group, Inc. static spray test 12/06
	e. Boom Pressure (pounds/square inch)	psi	55	1	20-60	Dynamic Aviation Group, Inc. static spray test 12/06
2	Transit Speed at Altitude From Base to Staging Airport	knots feet	185 20,000	1	150-200 12-20,000	Dynamic Aviation Group, Inc. field test Nov-06
3	Transit Speed at Altitude Staging Airport to/from spill	knots feet	185 1,500	1	150-200 1,500	Dynamic Aviation Group, Inc. field test Nov 06

1 Revised April 18, 2007

Figure 26A. Dispersant Aircraft Capability Form for Beechcraft King Air

4	Dispersant Spraying Reposition Speed	knots	150	1	130-185	Dynamic Aviation Group, Inc. field test Nov 06
5	Time to Fully Load Dispersant Tank	min	10	1	10-20	Dynamic Aviation Group, Inc. timed exercise
6	Time to Fully Load Fuel Tanks	min	12	1	10-30	Dynamic Aviation Group, Inc. operator exercise
7	Load Dispersant & Fuel simultaneously (Yes/No)	-----	YES	1	YES	Dynamic Aviation Group, operator See Other Comments below *5-*6
8	Time to Make U-turn (Turn 180 degrees)	min	1.00	1	1.0-3.0	Dynamic Aviation Group, Inc. operator measured exercise flight paths
9*	Dispersant Payload Maximum	gal	425	1	0-425	Dynamic Aviation Group, Inc. operator
10	Fuel with maximum dispersant payload	lbs	895	1	895	Dynamic Aviation Group, Inc. operator
11	Approach Distance for spraying	nm	.75	1	0.3-1.5	Dynamic Aviation Group, Inc. operator measured exercise paths Nov 06
12	Departure Distance for spraying	nm	.75	1	0.3-1.5	Dynamic Aviation Group, Inc. operator measured exercise paths Nov 06
13	Taxi Time Take-Off	min	3	2	3-15	Dynamic Aviation Group, Inc. operator
14	Taxi Time Landing	min	3	2	3-15	Dynamic Aviation Group, Inc. operator
15	On-site Check-In/Safety Time	min	10	2	5-15	Dynamic Aviation Group, Inc. operator
	CASCADE PARAMETERS*					
16 *	Take-off with **Maximum Payload and Maximum Take-off Weight** (assume no wind and VFR fuel reserve)					
	a. Maximum Flight Time	hours	1.2	2	1.2	Dynamic Aviation Group, Inc. operator
	b. Maximum Flight Range	nm	222	2	192-240	Dynamic Aviation Group, Inc. operator
	c. Optimal Altitude	feet	20,000	2	20,000	Dynamic Aviation Group, Inc. operator
	d. True Air Speed	knots	185	2	160-200	Dynamic Aviation Group, Inc. operator
	e. Fuel Consumption	lbs/ hour	500	2	500-520	Dynamic Aviation Group, Inc. operator
17 *	Take-Off with **Maximum Fuel and No Payload** (assume no wind and VFR fuel reserve)					
	a. Maximum Flight Time	hours	4.3	2	4.3	Dynamic Aviation Group, Inc. operator
	b. Maximum Flight Range	nm	796	2	688-860	Dynamic Aviation Group, Inc. operator
	c. Optimal Altitude	feet	20,000	2	20,000	Dynamic Aviation Group, Inc. operator
	d. True Air Speed	knots	185	2	160-200	Dynamic Aviation Group, Inc. operator
	e. Fuel Consumption	lbs/ hour	500	2	500-520	Dynamic Aviation Group, Inc. operator
18	Staging area briefing	min	45	2	30-60	Exercise observation

2 Revised April 18, 2007

Figure 26B. Dispersant Aircraft Capability Form for Beechcraft King Air

	AIRPORT PARAMETERS					
19	Runway length - **Minimum** (For take-off at maximum gross weight assuming sea level, 90° F, no wind using a balanced field concept, i.e., go, no go speed)	feet	4,200	2	4,200	Dynamic Aviation Group, Inc. operator
20	Runway weight restrictions for maximum aircraft weight	lbs	11,220	1	11,220	**Dynamic Aviation Group, Inc. operator. Aircraft maximum weight**
	OTHER COMMENTS					
5*-6*	The time to load dispersants and fuel are stand alone times independent of each other. If item 7 indicates that fuel and dispersants can be loaded simultaneously, then the longer of fuel or dispersant load time is used in the capability calculations. If item 7 indicates fuel and dispersants can NOT be loaded simultaneously, then the times are added together to calculate the aircrafts capability. To load simultaneously depends upon the airport, aircraft, and support crew. The loading times depend upon the loading system i.e., 5,000 tank truck, 55 gallon drums or other means and the pumping system used. The time shown in items 5 and 6 is for loading from a tank truck which is standing by ready to commence loading when the aircraft comes to a stop in the loading area, i.e. the fastest loading time possible.					
9*	Maximum payload depends upon the weight of the dispersant per gallon. The calculations provided are for the heaviest dispersant, Corexit 9527, weighing 8.5 lbs/gallon. For this aircraft the payload can be reduced from 425 gallons and the fuel increased to extend the range of the aircraft. A loading chart for the aircraft is provided to the template as an attachment.					
*	**Cascade Parameters:** The aircraft's calculated capability when cascading uses the same fuel loading and taxi times for dispersant operations as listed in items 6, 13 and 14.					
16 *	Calculations are based on empty weight plus crew of 6,768 pounds and gross take-off weight of 11,220 pounds and using Visual Flight Rules requiring a 30 minute fuel reserve. Payload, endurance and range depend upon the weight of the dispersant per gallon. The calculations provided are for the heaviest dispersant, Corexit 9527, weighing 8.5 lbs/gallon. Increased range may be obtained by increasing fuel and decreasing dispersant to the maximum fuel load for the aircraft of 2479 lbs. Flight hours for the aircraft can be estimated for planning purposes from the loading chart attached to this template, which also includes calculations for Corexit 9500. Pilots of the aircraft will determine payloads versus fuel loading based on the specific aircraft being used, the temperature, humidity, airport altitude, weather conditions enroute, etc.					
17 *	The aircraft with a maximum fuel load of 370 gallons (2479 lbs) can carry a payload of 1955 lbs of dispersant and retain the same range and speeds as for no payload.					

Figure 26C. Dispersant Aircraft Capability Form for Beechcraft King Air

BEECHCRAFT KING AIR (BE-90A)

LOADING CHART

Fuel Loaded before Engine Starts*		Dispersant			Aircraft Gross Weight At Takeoff	Endurance to Reserve	
Gallons	LBS	Corexit 9500 (7.9 #/g) Gallons	Corexit 9527 (8.5 #/g) Gallons	LBS	LBS	VFR 30 min reserve	IFR 45 min reserve
370	2479	160	149	1264	10,456	4.32	4.07
370	2479	170	158	1343	10,535	4.32	4.07
370	2479	180	167	1422	10,614	4.32	4.07
370	2479	190	177	1501	10,693	4.32	4.07
370	2479	200	186	1580	10,772	4.32	4.07
370	2479	210	195	1659	10,851	4.32	4.07
370	2479	220	204	1738	10,930	4.32	4.07
370	2479	230	214	1817	11,009	4.32	4.07
370	2479	240	223	1896	11,088	4.32	4.07
370	2479	250	232	1975	11,167	4.32	4.07
370	2479	257	239	2028	11,220	4.32	4.07
366	2453	260	242	2054	11,220	4.27	4.02
354	2374	270	251	2133	11,220	4.11	3.86
343	2295	280	260	2212	11,220	3.96	3.71
331	2216	290	270	2291	11,220	3.80	3.55
319	2137	300	279	2370	11,220	3.64	3.39
307	2058	310	288	2449	11,220	3.49	3.24
295	1979	320	297	2528	11,220	3.33	3.08
284	1900	330	307	2607	11,220	3.17	2.92
272	1821	340	316	2686	11,220	3.01	2.76
260	1742	350	325	2765	11,220	2.86	2.61
248	1663	360	335	2844	11,220	2.70	2.45
236	1584	370	344	2923	11,220	2.54	2.29
225	1505	380	353	3002	11,220	2.39	2.14
213	1426	390	362	3081	11,220	2.23	1.98
201	1347	400	372	3160	11,220	2.07	1.82
189	1268	410	381	3239	11,220	1.92	1.67
177	1189	420	390	3318	11,220	1.76	1.51
172	1150	425	395	3358	11,220	1.68	1.43
165	1108	----	400	3400	11,220	1.60	1.35
153	1022	----	410	3485	11,220	1.43	1.18
140	937	----	420	3570	11,220	1.26	1.01
134	895	----	425	3613	11,220	1.17	.92

* Estimated that aircraft will use 55 pounds of fuel for run-up to taking off so that at takeoff aircraft will have 55 pounds less fuel than shown here. At 6.7 pounds of fuel per gallon this equates to 8.2 gallons of fuel. The flight times are based on the aircraft using 502 lbs of fuel per hour of the fuel available at takeoff.

Figure 26D. Dispersant Aircraft Capability Form for Beechcraft King Air

Vessel Platform Inputs

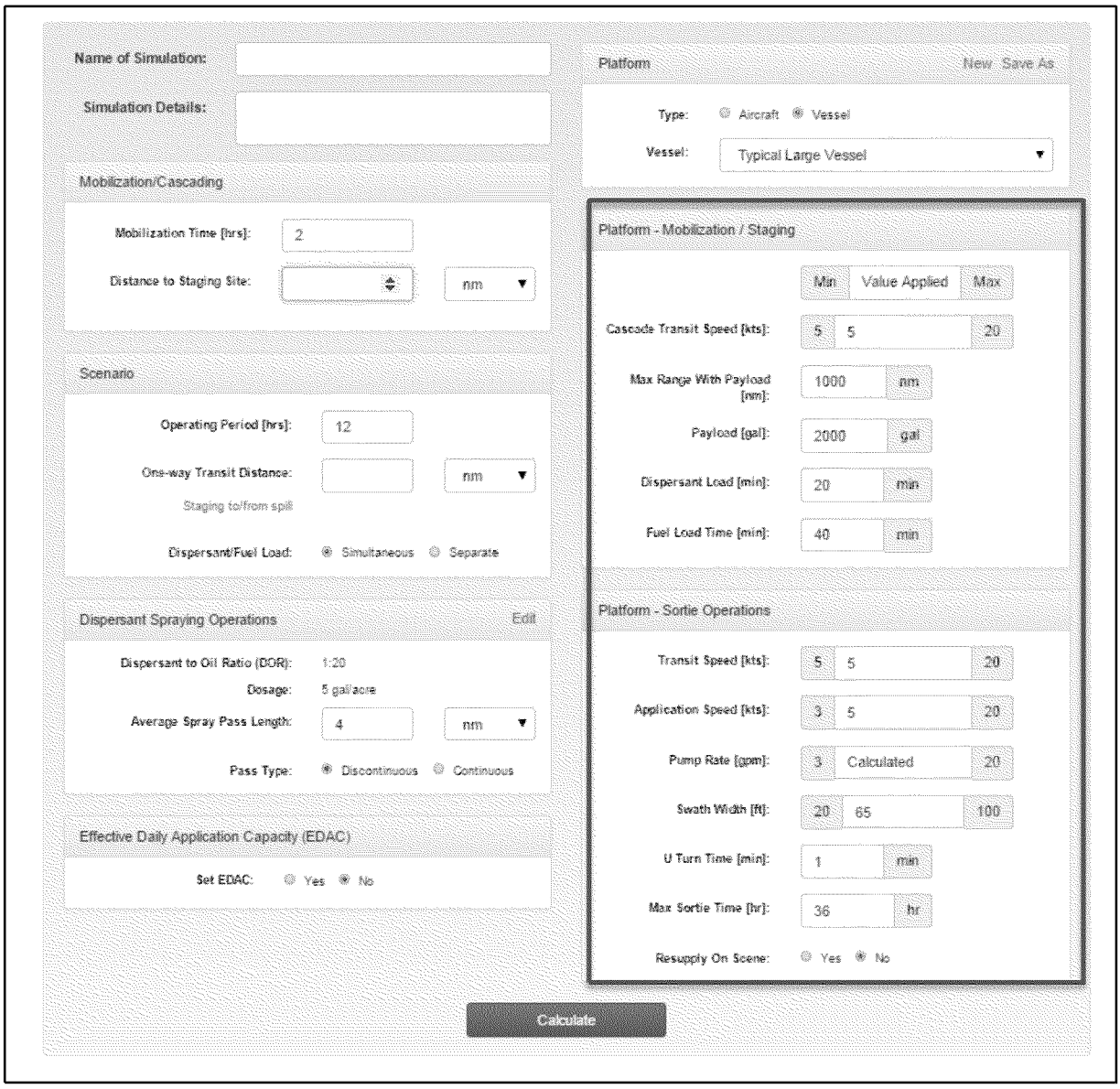


Figure 27. Vessel Platform Input.

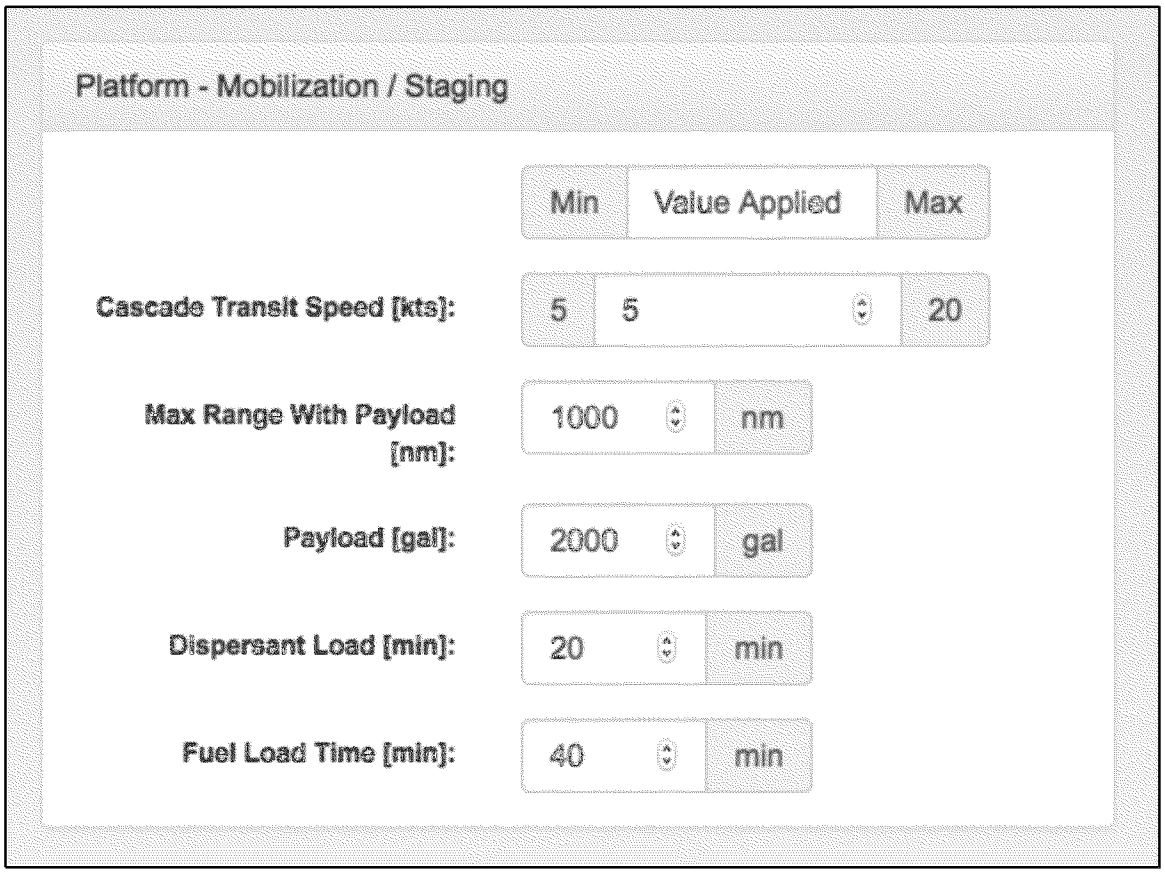


Figure 28. Vessel Platform – Mobilization/Staging Group.

Mobilization/Staging

Cascade Transit Speed: The speed at which the vessel transits while cascading from out of area. Three values are displayed: the minimum Cascade Transit Speed, the default value used for calculations, and the maximum allowable Cascade Transit Speed.

Max Range: This is the maximum range this vessel has with maximum fuel load

Dispersant Payload: The maximum volume of dispersant the selected vessel can carry.

Dispersant Load Time: The time necessary for a complete dispersant load.

Fuel Load Time: The time necessary for a complete fuel load.

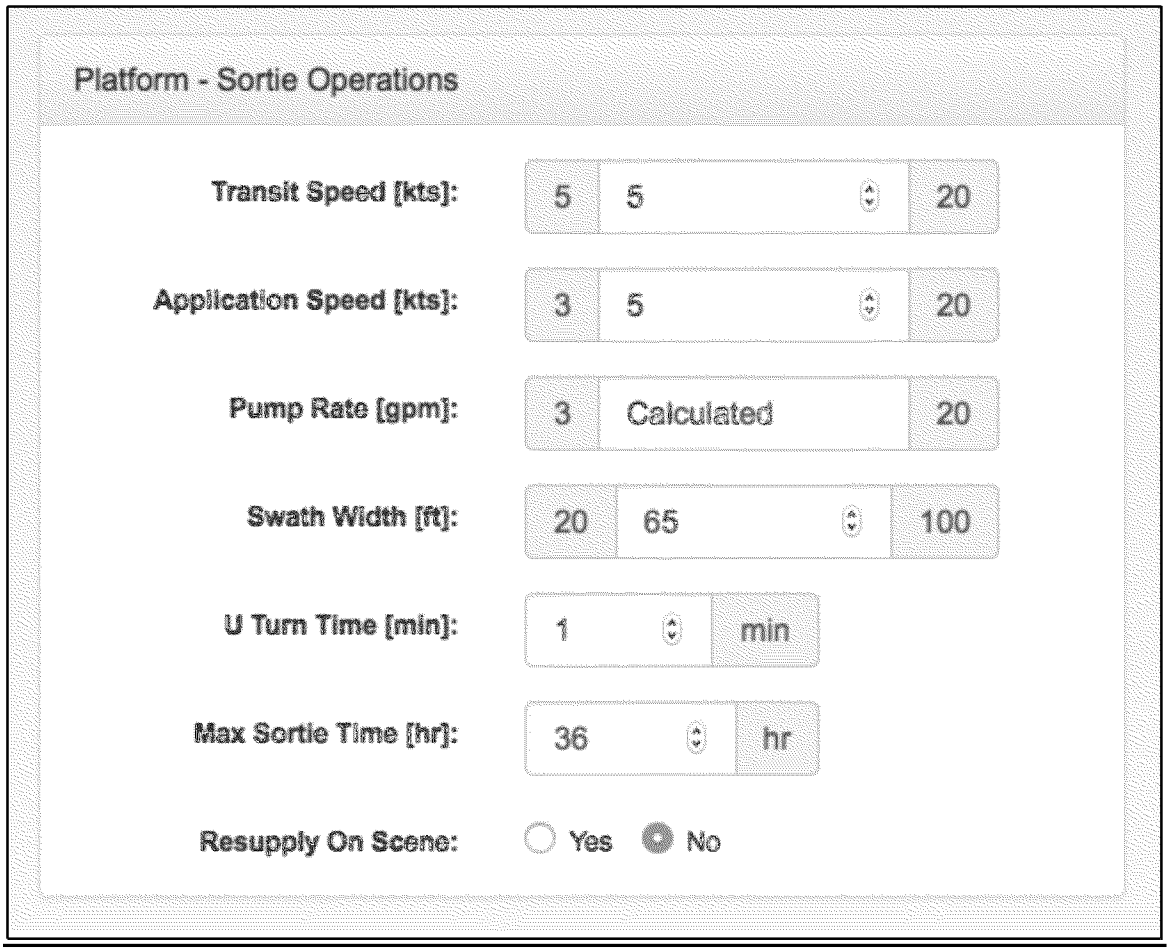


Figure 29. Vessel Platform - Sortie Operations.

Sortie Operations

Transit Speed: Three values are displayed; the minimum Transit Speed, the value used for calculations, and the maximum Transit Speed.

Application Speed: The vessel speed while spraying dispersant.

Pump Rate: The calculated Pump Rate necessary to deliver the required dosage. The EDSP checks to insure that the calculated pump rate is between the maximum and minimum pump rates and assumes that the selected platform can deliver the calculated pump rate.

Swath Width: The width of the dispersant spray onto the oil slick.

U Turn Time: The time necessary for the vessel to make a U-turn.

Max Sortie Time: The maximum time the vessel can operate, which is a function of the dispersant load, the onboard berthing and messing capacities, and the maximum range.

Resupply On Scene: The default setting is "No". When "Resupply On Scene" is set to "No", if a vessel has less than one half of its dispersant payload left onboard when the OP ends, the platform will immediately transit back to the staging area to resupply. However, if the vessel has more than one half of its payload remaining onboard at the end of the OP, the platform will stay at the spill site between operating periods and will commence spraying the remainder of the payload at the beginning of the next operating period. Figure 29 below shows a vessel timeline with Resupply On Scene set to "No."

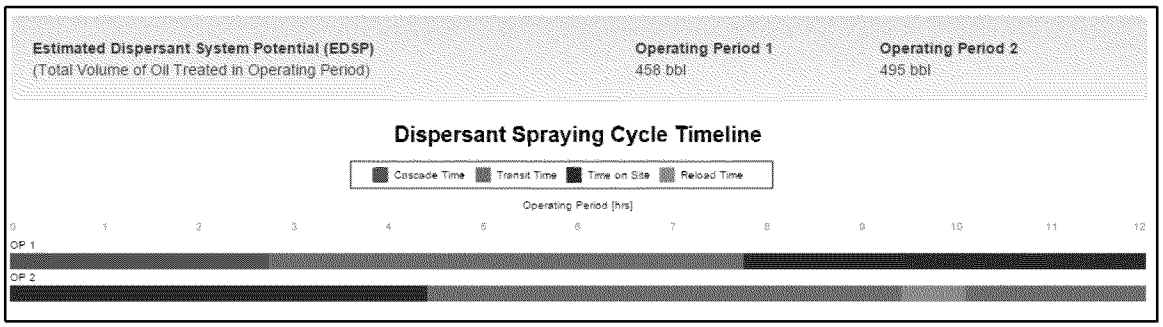


Figure 30. Vessel Timeline with Resupply On Scene set to "No."

When "Yes" is selected the vessel will be resupplied with dispersant on-scene, either when the payload of dispersant is completely applied during spraying operations, or when the operating period is completed, whichever occurs first. Figure 30 shows the same vessel timeline with Resupply On Scene set to "Yes."

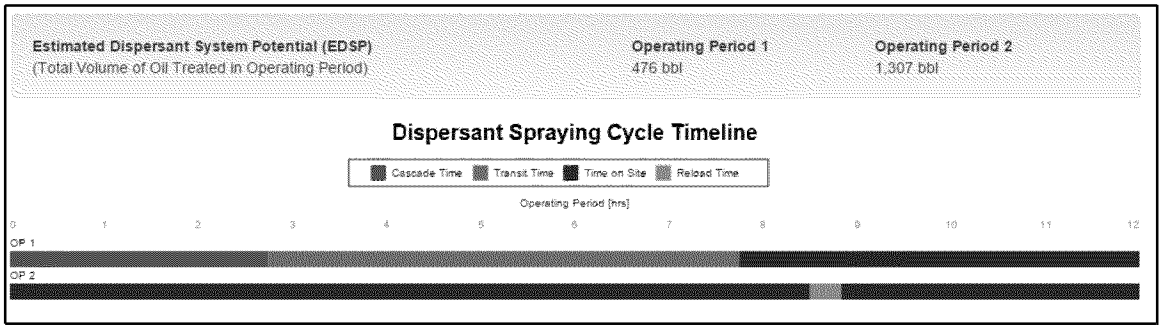


Figure 31. Vessel Timeline with Resupply On Scene set to "Yes."

Changing Platform Values:

Values for the platform parameters can be changed (some values will have a minimum and maximum value that will bound the lower and upper limits of possible alternative values that may be entered). When this is done the box around the changed value will be outlined in orange to indicate a default value has changed. Below the Transit Speed [kts] and Departure [nm] have been changed and the box around those two values has turned orange to highlight that these values have been changed.

Figure 32. Example of Platform Value Change

Creating Spray Application Platforms with Alternative Values for Inputs

Users can create customized platforms with alternative input values via two different methods:

1) Users may create a new customized platform completely from scratch by:

    a. Clicking on "New" in the Platform Menu Box
    b. Selecting "Aircraft" or "Vessel" in the "New Platform" Dialogue Box and clicking on "Save"
    c. Entering in the platform name and its input values in the "New Platform" Menu and clicking on "Save"

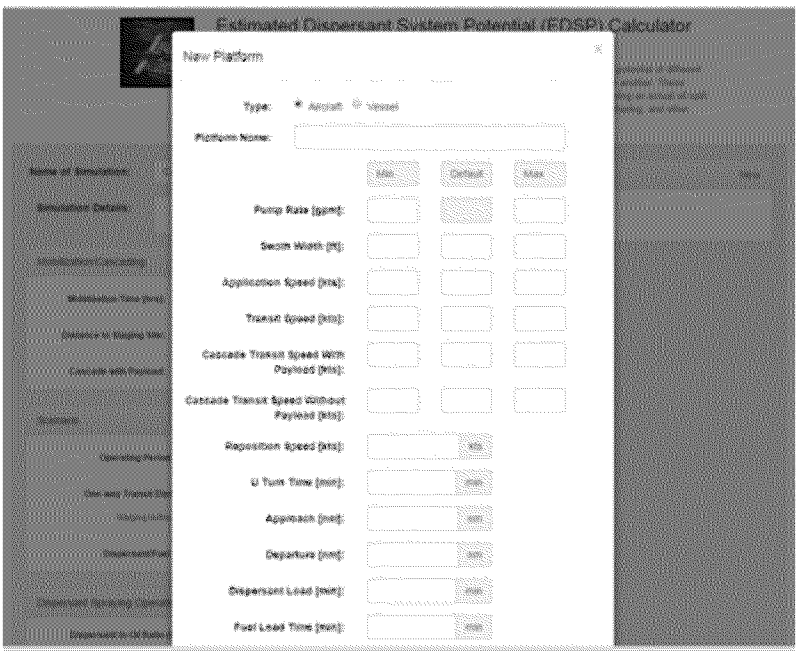


Figure 33. New Aircraft platform template.

2) Users may also modify one or more input parameters for an existing platform and save this configuration as a new platform by:

    a. Modifying the existing input parameters with the desired alternative values and then clicking on "Save As" in the Platform Menu Box. In Figure 33 below, the payload was changed from 3250 to 4000 gallons.

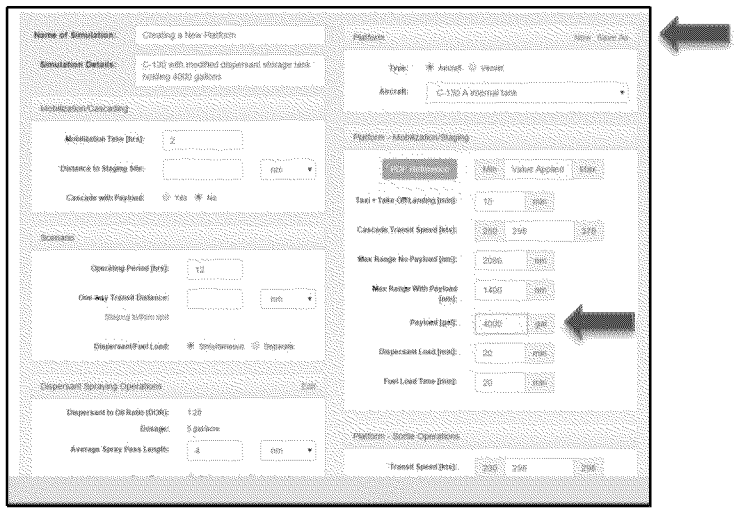


Figure 34. Aircraft Payload Change

b. Edit the Platform Name, confirm the alternative values are entered into the “Edit Platform” Menu, and then click on “Save”

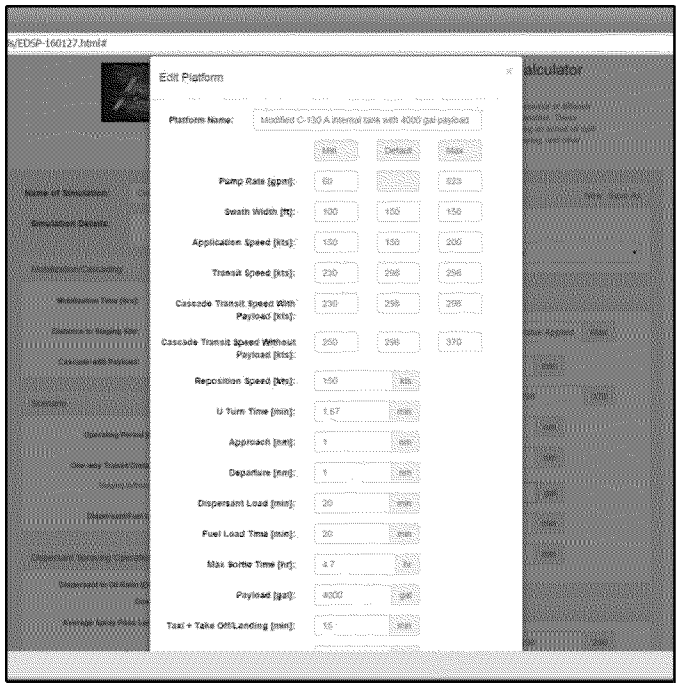


Figure 35. Saving Aircraft Platform Change

Regardless of which method you use, once you have saved your platform with the alternative input values, it will now appear in your platform drop down menu/picklist for use in your locally maintained version of the EDSP Calculator.

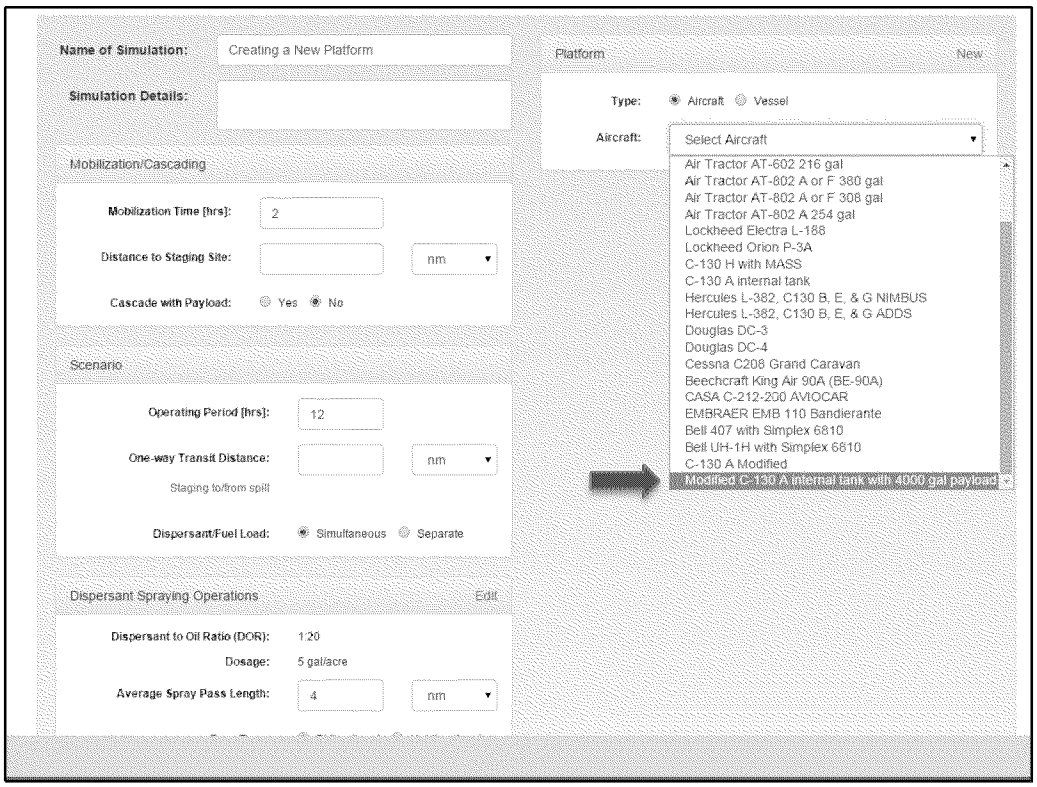


Figure 36. New Aircraft Platform Added to Selection List

Note: If you modify existing input values for a platform, and do not save those changes as a custom platform, whenever you change platform selections, all modified input values will be automatically reset to their default values.

4. EDSP Calculator Outputs

EDSP Calculator results are output in the form of both graphics and tabular data. The graphical data is presented in the form of summary data and a Dispersant Spraying Cycle Timeline during each Operating Period.

Estimated Dispersant System Potential (EDSP) (Total Volume of Oil Treated in Operating Period)	Operating Period 1	Operating Period 2
	7,143 bbl	9,524 bbl

Figure 37. Summary Data.

The Dispersant Spraying Cycle Timeline depicts mobilization/cascading, transit, dispersant spraying and reloading periods for Operating Period 1 on an hourly timescale. Operating Period 2 depicts the first full day on scene.

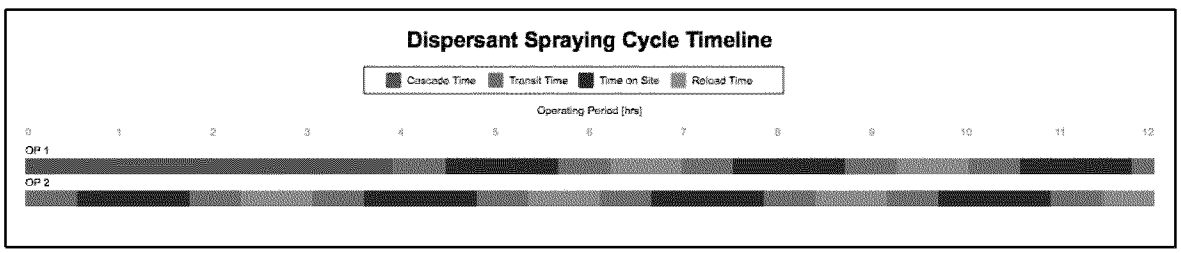


Figure 38. Dispersant Spraying Cycle Timeline.

The following figure shows the tabular data outputs, which present additional planning details relating to cascade time, and dispersant spraying related aspects of the system.

Results for Each Operating Period	OP 1	OP 2
Cascade Time	3.92 hrs	0 hrs
Time On Scene to Commence Spray Operations for OP 1 (Cascade Time + One Way Transit Time)	4.5 hrs	N/A
Adjusted OP Time	8.08 hrs	12 hrs
Payload Deliveries	3	4
Dispersant Applied	15000 gal	20000 gal
Total Area Coverage	2999 acres	3999 acres
Oil Treated (EDSP)	7143 bbl	9524 bbl
Results per Sortie for a Complete Payload Application		
One-way Transit Time (Including Taxi + Takeoff/Landing for Aircraft)	33 min	
Calculated Pump Rate	253 gpm	
Spray Time/Pass	1.7 min	
# of Passes/Sortie	11.9	
Spray Time/Sortie	19.8 min	
Total Time/Sortie	138 min	
Areal Coverage Rate	50.6 acres/min	
Area Covered/Sortie	1000 acres	

Figure 39. EDSP Tabular Outputs.

4a. Results for Each Operating Period

Cascade Time:

This includes Notification/Mobilization time, travel time to staging site and a 45 minute briefing period for both aircraft and vessels.

Time On Scene to Commence Spray Operations for OP1 (Cascade Time + One Way Transit Time):

This is the time necessary for the platform to arrive on scene and ready to spray, only in OP1.

Adjusted OP Time:

For OP 1, this is the Operating Period time minus the Cascade time.

Payload Deliveries:

Total number of payload deliveries, including any partial payload deliveries completed by the end of the operating period.

Dispersant Applied:

Total number of gallons of dispersant applied in the operating period.

Total Area Coverage:

Total area sprayed in the operating period.

Oil Treated (EDSP):

Amount of oil assumed to be treated based on the dispersant to oil ratio.

4b. Results per Sortie for a Complete Payload Application

One-way Transit Time (including Taxi + Takeoff/Landing for Aircraft):

Amount of time needed to transit from the staging area to the spill site.

Calculated Pump Rate:

This is the pump rate necessary to achieve the desired dosage.

Spray Time/Pass:

Amount of time actually spraying on each pass over the oil slick.

# of Passes/Sortie:

The number of passes over the oil slick necessary to apply one payload of dispersant.

Spray Time/Sortie:

The total amount of time actually spraying during each payload delivery.

Total Time/Sortie:

Includes spraying time, repositioning time, and transit time.

Areal Coverage Rate:

Number of acres covered for each minute of spraying time.

Area Covered/Sortie:

Total number of acres sprayed per payload.

4c. Aircraft Sortie Timeline Components

The following graphic illustrates all the components of an aircraft dispersant application operation that are included in the EDSP

EDSP Calculator Aircraft Timeline Components

Timeline color legend	
	Cascade Time
	Transit Time
	Time on Site
	Reload Time
	Total sortie time

		Timeline Component	Source
		Notification & Mobilization	Input
		Taxi & takeoff time	From platform data
		Cascade transit to staging time (includes any required fuel stops)	Calculated using Cascade Transit Speed
		Landing & taxi time	From platform data
		Initial orientation briefing, refueling & loading	45 min
One Sortie		Taxi & takeoff time	From platform data
		Transit to operational area	Calculated using Transit Speed
		On-site spotter briefing	10 min
		Approach time (Pass 1)	Calculated using Reposition Speed
		Spray time (Pass 1)	Calculated using Application Speed
		Departure time (Pass 1)	Calculated using Reposition Speed
		U-Turn time (Pass 1)	From platform data
		Approach time (Pass 2)	Calculated using Reposition Speed
		Spray time (Pass 2)	Calculated using Application Speed
		Departure time (Pass 2)	Calculated using Reposition Speed
		U-Turn time (Pass 2)	From platform data
		Repeat spray pass sequence...	*Repeat ...*
		Approach time (Last pass)	Calculated using Reposition Speed
		Spray time (Last pass)	Calculated using Application Speed
		Departure time (Last pass)	Calculated using Reposition Speed
		Transit to staging area	Calculated using Transit Speed
		Landing & taxi time	From platform data
		Loading fuel & dispersant time (in parallel or serial)	From platform data
Next Sortie		Taxi & takeoff time	From platform data
		Transit to operational area	Calculated using Transit Speed
		On-site spotter briefing	10 min
		Approach time (Pass 1)	Calculated using Reposition Speed
		Spray time (Pass 1)	Calculated using Application Speed
		Departure time (Pass 1)	Calculated using Reposition Speed
		U-Turn time (Pass 1)	From platform data
		Approach time (Pass 2)	Calculated using Reposition Speed
		Spray time (Pass 2)	Calculated using Application Speed
		Departure time (Pass 2)	Calculated using Reposition Speed
		U-Turn time (Pass 2)	From platform data
		Repeat for other sorties to end of Operating Period...	*Repeat...*

Figure 40. Aircraft Sortie Timeline Components

4c. Results when using the EDAC button

When the EDAC button is set to “Yes”, the Calculator estimates the volume of dispersants applied in the “Results for Each Operating Period” and labels this data as “Computed EDAC”.

Estimated Dispersant System Potential (EDSP) (Total Volume of Oil Treated in Operating Period)	Operating Period 1	Operating Period 2
	4,643 bbl	7,738 bbl

Dispersant Spraying Cycle Timeline

Results for Each Operating Period	OP 1	OP 2
Cascade Time	4.25 hrs	0 hrs
Time On Scene to Commence Spray Operations for OP 1 (Cascade Time + One Way Transit Time)	5 hrs	N/A
Adjusted OP Time	7.75 hrs	12 hrs
Payload Deliveries	3	5
Dispersant Applied (Computed EDAC)	9750 gal	16250 gal
Total Area Coverage	1947 acres	3245 acres
Oil Treated (EDSP)	4643 bbl	7738 bbl

Figure 41. Example of EDAC Output

5. Using the EDSP Calculator

Starting the EDSP Calculator

You can access the EDSP Calculator by double-clicking on the EDSP-160225.html file (date portion of file name will change as new builds are available), which will open up the Calculator in your default web browser.

5a. Calculating EDSP using an Aircraft

Entering input values

- In the **Name of Simulation** box, enter the identifier for this burn system configuration. This example uses "Spray 1".
- The **Simulation Details** box can be used to enter descriptive information about the dispersant system configuration.

Mobilization/Cascading Group:

- The **Mobilization Time [hrs]** box auto fills with 2 hrs, but can be changed.
- Set **Distance to Staging Site** to 200 nm.
- The **Cascade with Payload** default is "No", but can be changed.

Scenario Group:

- The **Operating Period** is auto set at 12 hrs, but can be changed.
- Enter 75 in the **One-way Transit Distance [nm]** box.
- Under **Dispersant/Fuel Load** select the "Simultaneous" radio button.

Platform Group:

- Select the Aircraft radio button.
- Select the Hercules L-382, C130 B, E, & G ADDS aircraft.

The Calculator input screen should now look like this:

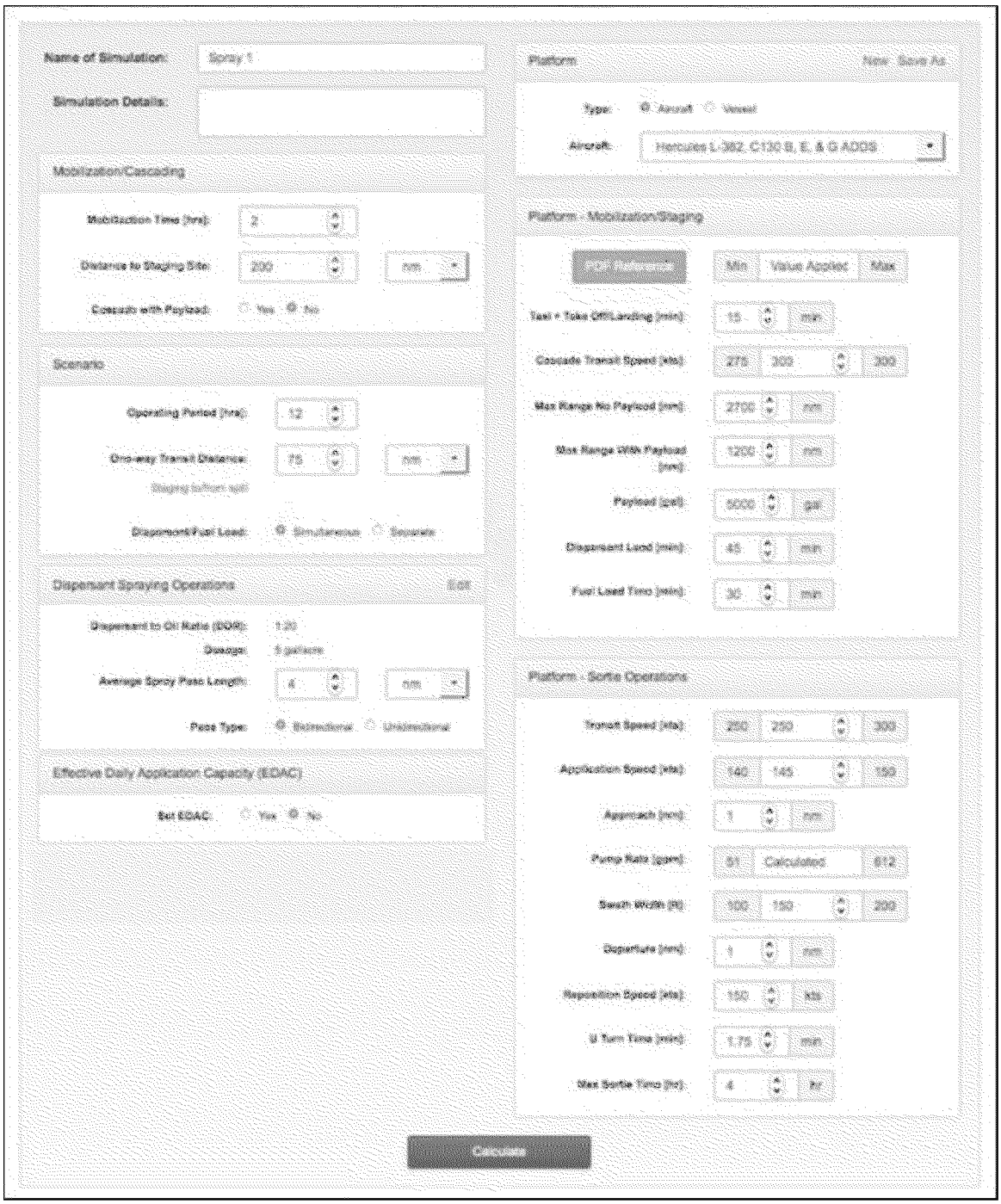


Figure 42. EDSP Input Screen for Aircraft.

Click **Calculate**. The EDSP Outputs for this dispersant application system configuration will then be displayed.

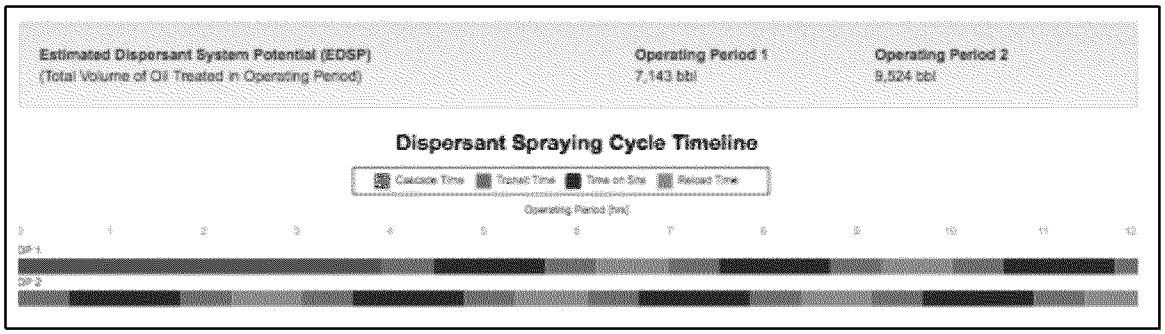


Figure 43. EDSP Timeline Outputs for Spray 1.

Results for Each Operating Period	OP 1	OP 2
Cascade Time	3.92 hrs	0 hrs
Time On Scene to Commence Spray Operations for OP 1 (Cascade Time + One Way Transit Time)	4.5 hrs	N/A
Adjusted OP Time	8.08 hrs	12 hrs
Payload Deliveries	3	4
Dispersant Applied	15000 gal	20000 gal
Total Area Coverage	2999 acres	3999 acres
Oil Treated (EDSP)	7143 bbl	9524 bbl
Results per Sortie for a Complete Payload Application		
One-way Transit Time (Including Taxi + Takeoff/Landing for Aircraft)	33 min	
Calculated Pump Rate	253 gpm	
Spray Time/Pass	1.7 min	
# of Passes/Sortie	11.9	
Spray Time/Sortie	19.8 min	
Total Time/Sortie	138 min	
Areal Coverage Rate	50.6 acres/min	
Area Covered/Sortie	1000 acres	

Figure 44. EDSP Tabular Output for Spray 1.

For this dispersant spraying system configuration and scenario, the calculated EDSP value is 7143 barrels of oil for OP1 and 9524 barrels of oil for OP2. During the twelve hour Operating Period, you can expect this system, as configured, to deliver three payloads in OP1 and four in OP2.

5b. Variations in Aircraft Spray 1 Scenario

Changing Dispersant Spraying Options Inputs:

To change the Dispersant Spraying Options select the "Edit" link in the upper right hand corner to make the "Dosage" window pop up. The Dosage can be changed with Input Dosage Directly (default) or Estimate Dosage from Slick Description. If Estimate Dosage from Slick Description is selected, enter the estimate the volume of oil spilled, the estimated area covered by the oil slick, and the estimated percentage coverage of the slick. The resultant average thickness will be used with the DOR to calculate the necessary dosage to treat the oil slick.

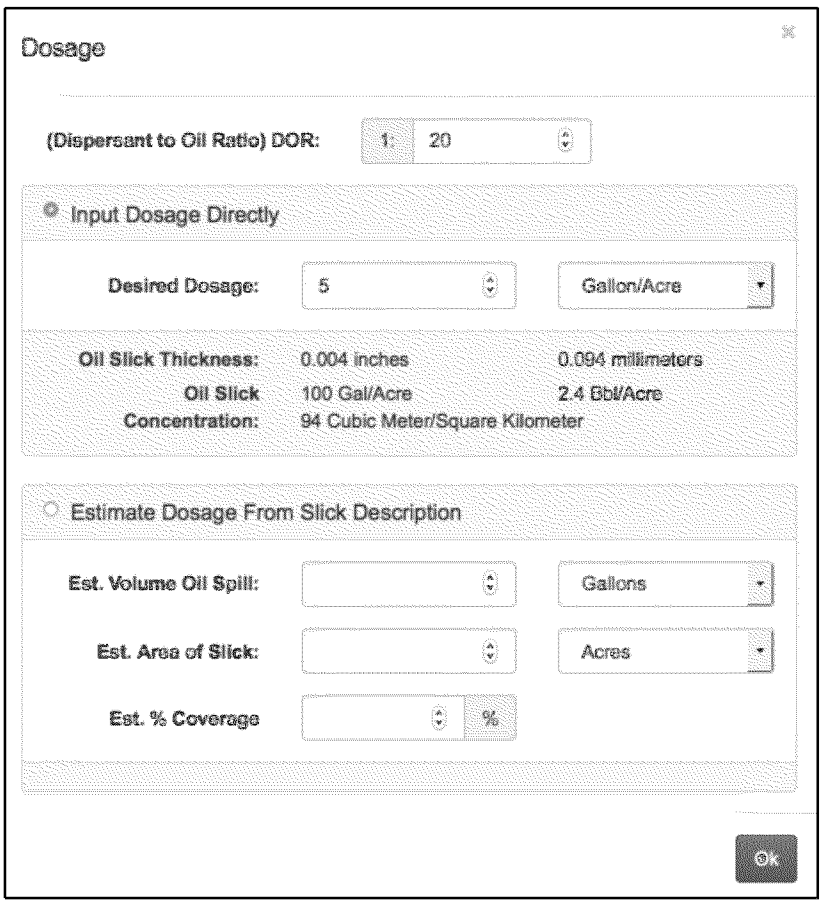


Figure 45. DOR/Dosage Edit.

Important Note. The Dispersant Aircraft Capability Forms do not contain explicit dosage capability values or ranges. Every aircraft platform in the EDSP database can deliver the regulatory dosage of 5 gallons of dispersant for each acre sprayed. Other dosage values used in the EDSP may not be actually achievable for a specific platform and should be verified with the platform operator before use.

There will be cases where the desired dosage is too low to be delivered by the specified platform. In that instance an Error box will be displayed, as seen below:

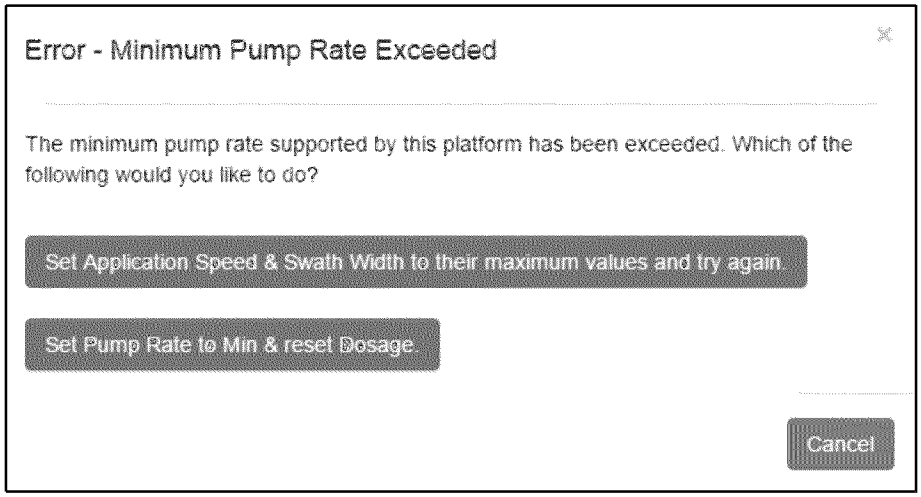


Figure 46. Minimum Pump Rate Error.

In this case the choices are, 1) Set the platform application speed and swath width to their maximum values and try again, or, 2) Set the platform's pump rate to its minimum value and the achievable dosage value will be set.

Likewise, when a dosage value is specified that is higher than the platform can deliver, the following error dialogue box will be displayed:

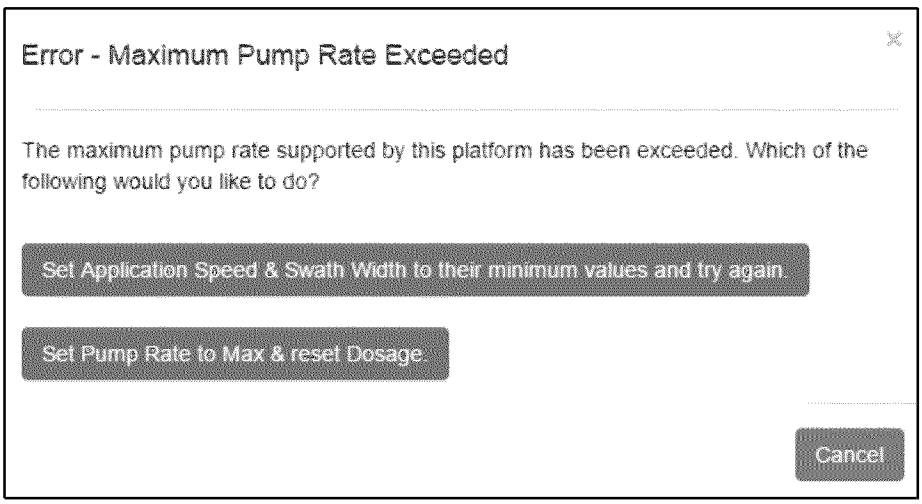


Figure 47. Maximum Pump Rate Error.

The choices are 1), to reset the Application Speed and Swath Width to their minimum values and try again, or 2) Set Pump Rate to its maximum value and the achievable Dosage will be calculated. Multiple spray passes from the platform over the oil slick may be necessary if the oil is too thick.

Cascade and Time On Scene Logic for Aircraft in the EDSP:

- **On Scene In-between OP1 and OP2:** If the Time On Scene to Conduct Spray Operations (TOSCO) of a dispersant aircraft will be greater than one full Operating Period but less than 24 hours, then Operating Period 1 will show solely as Cascade Time and OP2 will begin with a transit to the spill and spray operation.
- **On Scene during OP2:** If TOSCO is greater than 24 hours but less than (24 hours + OP hours), OP1 and OP2 will display actual cascade and transit times and OP2 will correctly show when the aircraft arrives on scene to commence spray operations.
- **On Scene After the end of OP2**: If TOSCO is greater than (24 hours + OP hours), then OP2 will begin with a transit (i.e. the aircraft platform arrives on scene after the end of OP 2).

The following Figures illustrate a scenario where a C-130 is mobilized and cascaded from Arizona to a staging site on the North Slope of Alaska to apply dispersants on a spill offshore in the Chukchi Sea. This is an example of an aircraft arriving on scene between OP1 and OP2.

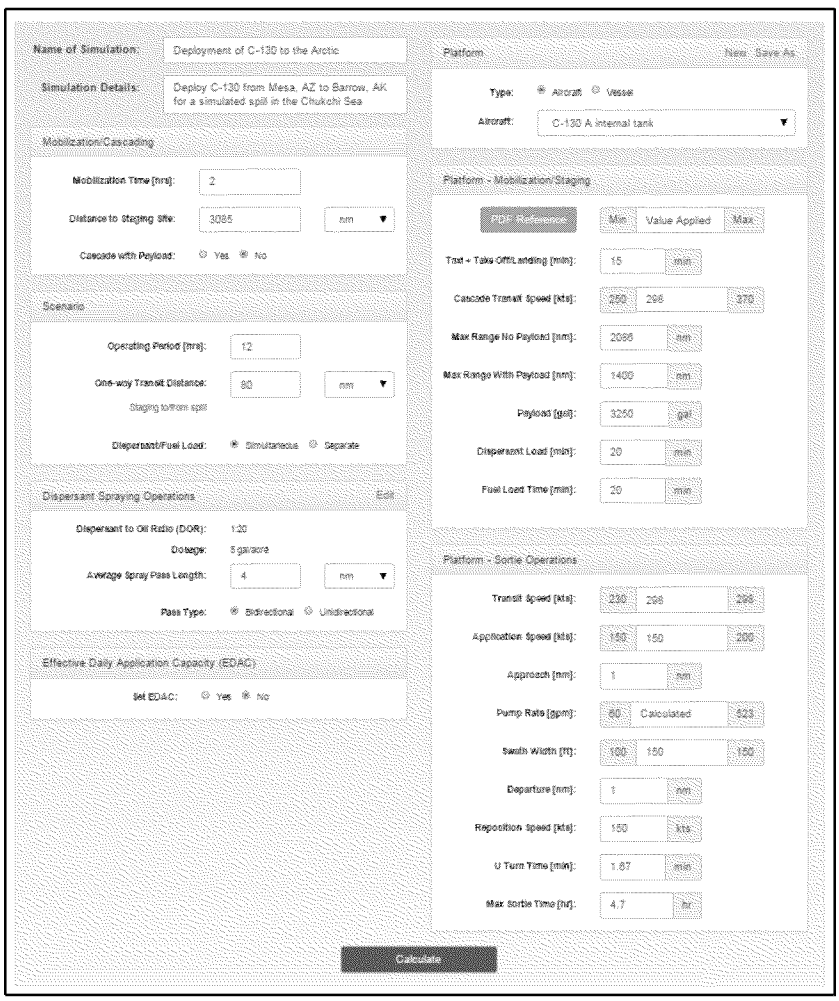
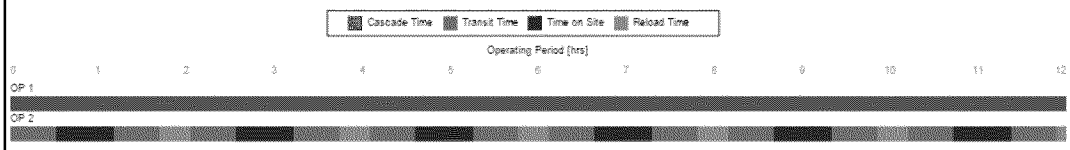


Figure 48. C-130 Arctic Cascade Scenario

Estimated Dispersant System Potential (EDSP) (Total Volume of Oil Treated in Operating Period)	Operating Period 1	Operating Period 2
	0 bbl	9,286 bbl

Dispersant Spraying Cycle Timeline



Results for Each Operating Period	OP 1	OP 2
Cascade Time	14.44 hrs	0 hrs
Time On Scene to Commence Spray Operations for OP 1 (Cascade Time + One Way Transit Time)	15 hrs	N/A
Adjusted OP Time		12 hrs
Payload Deliveries		6
Dispersant Applied		19500 gal
Total Area Coverage		3894 acres
Oil Treated (EDSP)		9286 bbl

Results per Sortie for a Complete Payload Application	
One-way Transit Time (Including Taxi + Takeoff/Landing for Aircraft)	31 min
Calculated Pump Rate	262 gpm
Spray Time/Pass	1.6 min
# of Passes/Sortie	7.8
Spray Time/Sortie	12.4 min
Total Time/Sortie	102.3 min
Areal Coverage Rate	52.3 acres/min
Area Covered/Sortie	649 acres

Figure 49. C-130 Arctic Cascade Scenario Results

The computed cascade time for this example as shown in the results table in the OP1 column is 14.4 hours. As a result, the timeline for OP 1 is made up entirely of Cascade Time. As the platform can be ready to begin spraying before the second OP begins (15 hours), the second OP begins with a transit to the spill site to begin spray operations. The platform is capable of delivering 6 payloads for a total of 19,500 gallons of dispersant applied (treating an estimated 9286 barrels of oil) during OP 2. These numbers reflect the EDAC/EDSP potential that the platform can achieve each day as the spill continues.

In the following example, a C-130 is cascaded in from an international location (5000 miles away) that also requires a 24 hour period for preparing the aircraft for mobilization. The following Figures demonstrate the cascading logic and Calculator outputs for a platform that would arrive on scene after the end of OP2.

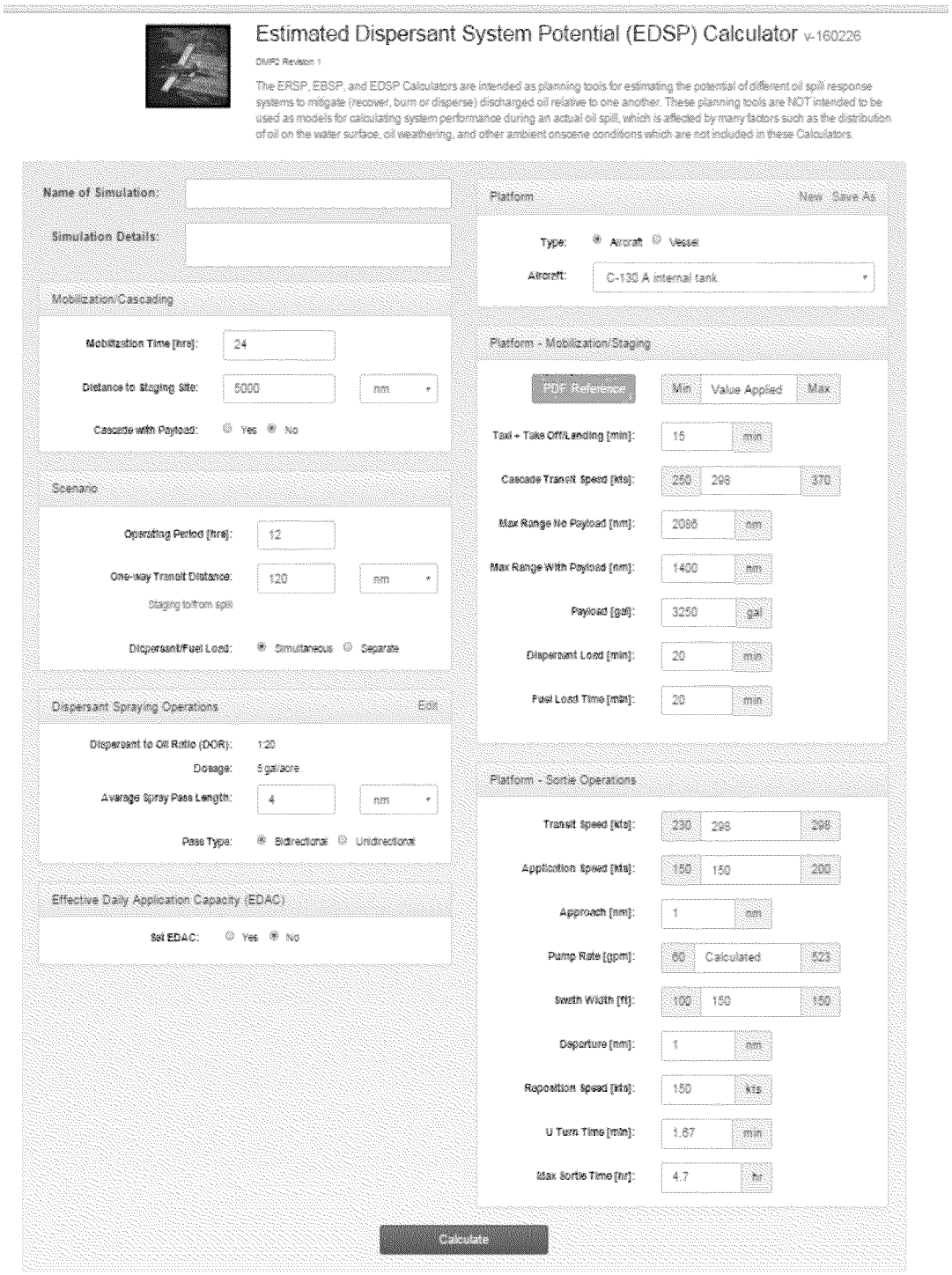
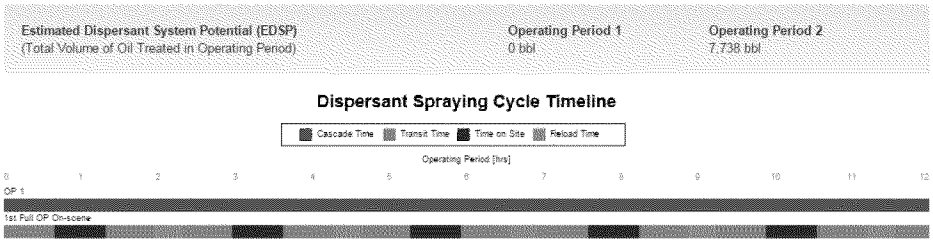


Figure 50. International Cascade Scenario Results



Results for Each Operating Period	OP 1	1st Full OP On-scene
Cascade Time	43.7 hrs	0 hrs
Time On Scene to Commence Spray Operations for OP 1 (Cascade Time + One Way Transit Time)	44.3 hrs	N/A
Adjusted OP Time	0 hrs	12 hrs
Payload Deliveries	0	5
Dispersant Applied	0 gal	16250 gal
Total Area Coverage	0 acres	3245 acres
Oil Treated (EDSP)	0 bbl	7738 bbl
Results per Sortie for a Complete Payload Application		
One-way Transit Time (Including Taxi + Takeoff/Landing for Aircraft)		39 min
Calculated Pump Rate		262 gpm
Spray Time/Pass		1.6 min
# of Passes/Sortie		7.8
Spray Time/Sortie		12.4 min
Total Time/Sortie		118.4 min
Areal Coverage Rate		52.3 acres/min
Area Covered/Sortie		649 acres

Figure 51. International Cascade Scenario Results

The computed cascade time for this example as shown in the results table in the OP1 column is 43.7 hours and the aircraft could be on scene as early as 44.3 hours. As a result, the timeline for OP 1 is made up entirely of Cascade Time. Notice the second bar in the timeline and the second column in the table are now labeled as "!st Full OP On Scene", and both reflect what the spray aircraft can achieve during a full operating period (where the platform begins the OP with a transit out to the spill site). As the platform could be on scene to begin spraying after 44.3 hours (and the third OP would begin at 48 hours), the second timeline bar and data column represent what the platform could provide during the third OP and each subsequent full OP afterwards. For this scenario, the platform is capable of delivering 5 payloads for a total of 16,2500 gallons of dispersant applied (treating an estimated 7738 barrels of oil) during the third OP.

Changing Cascade Speed with Payload:

For some aircraft platforms the cascade speed will change based on whether or not they are cascading with a dispersant payload. For example, the Air Tractor AT-502 B 170 gal aircraft has a Cascade Transit Speed of 122 kts when it cascades with no payload. However, when a full payload of dispersant is added, the Cascade Transit Speed decreases to 120 kts. The calculator takes these variations into consideration in the calculation of cascade time

5c. Calculating EDSP using a Vessel

Entering input values

- In the **Name of Simulation** box, enter the identifier for this burn system configuration. This example uses "Spray 2".
- The **Simulation Details** box can be used to enter descriptive information about the dispersant system configuration.

Mobilization/Cascading Group:

- The **Mobilization Time [hrs]** box auto fills with 2 hrs, but can be changed.
- Set **Distance to Staging Site** to 35 nm.

Scenario Group:

- The **Operating Period** is auto set at 12 hrs, but can be changed.
- Enter 15 in the **One-way Transit Distance [nm]** box.
- Under **Dispersant/Fuel Load** select the "Simultaneous" radio button.

Platform Group:

- Select the Vessel radio button.
- Select the Typical Large Vessel.
- Adjust any platform input values as necessary
- Select whether the vessel will be resupplied with dispersant on scene, or will have to transit back to the staging site to reload dispersants.

The Calculator input screen should now look like this:

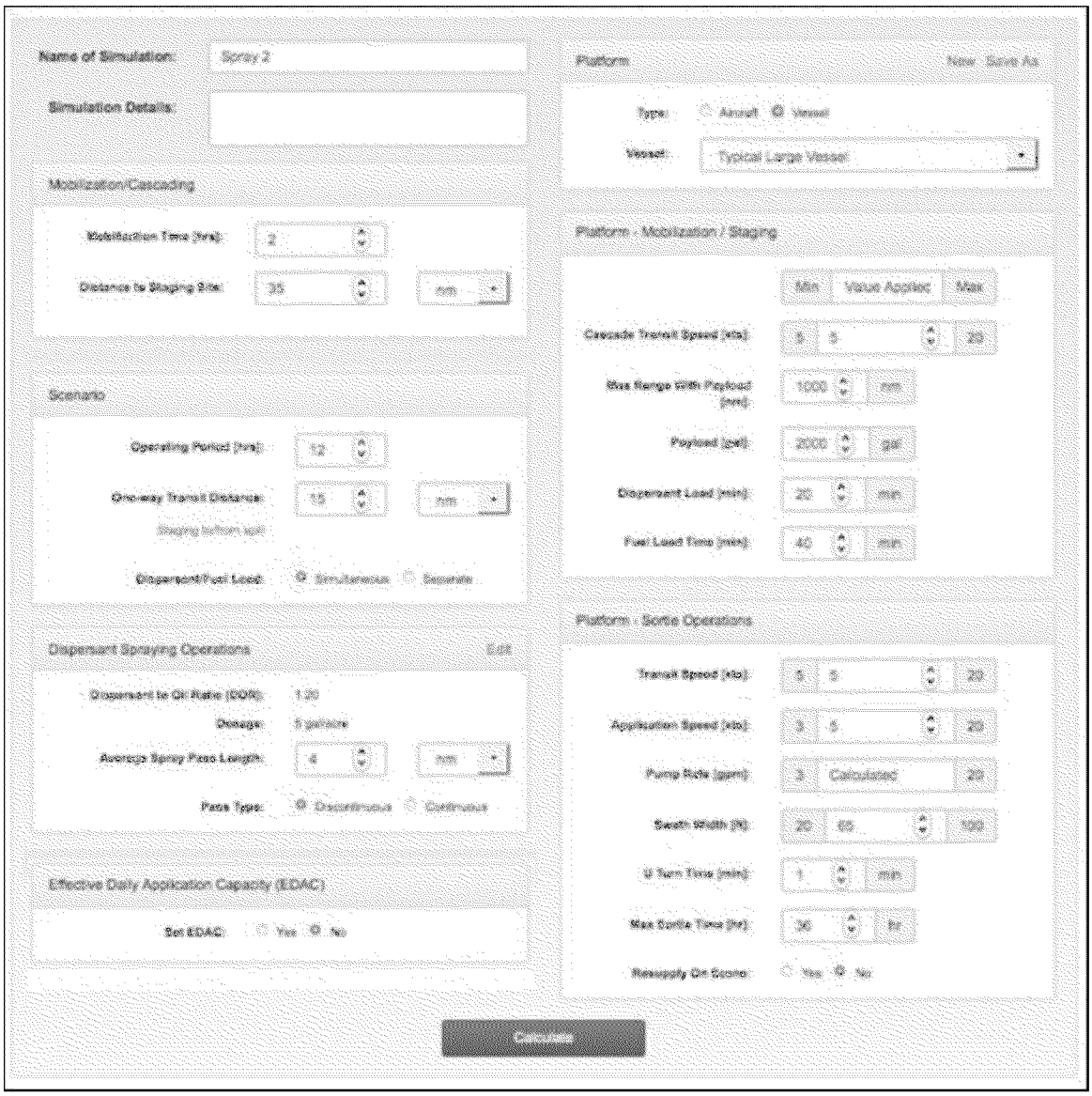


Figure 52. Vessel Input Screen.

Click **Calculate**. The EDSP Outputs for this dispersant application system configuration and scenario will then be displayed.

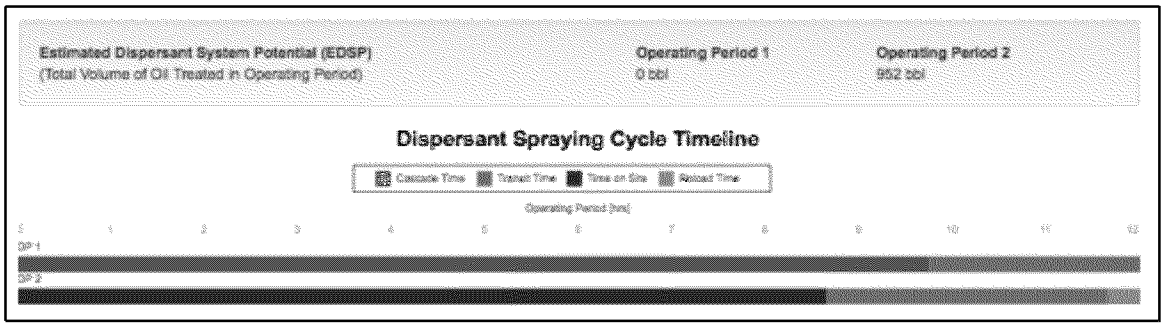


Figure 53. EDSP Graphical Outputs for Spray 2

Results for Each Operating Period	OP 1	OP 2
Cascade Time	9.75 hrs	0 hrs
Time On Scene to Commence Spray Operations for OP 1 (Cascade Time + One Way Transit Time)	12.8 hrs	N/A
Adjusted OP Time	2.25 hrs	12 hrs
Payload Deliveries	0	1
Dispersant Applied	0 gal	2000 gal
Total Area Coverage	0 acres	378 acres
Oil Treated (EDSP)	0 bbl	952 bbl
Results per Sortie for a Complete Payload Application		
One-way Transit Time (including Taxi + Takeoff/Landing for Aircraft)	180 min	
Calculated Pump Rate	4 gpm	
Spray Time/Pass	46 min	
# of Passes/Sortie	10.4	
Spray Time/Sortie	500 min	
Total Time/Sortie	870 min	
Areal Coverage Rate	0.8 acres/min	
Area Covered/Sortie	378 acres	

Figure 54. EDSP Tabular Output for Spray 2

For this dispersant spraying system configuration, the calculated EDSP value is 0 barrels of oil for OP1 because the vessel speed did not allow a spraying period in OP1 and 952 barrels of oil for OP2. During the twelve hour Operating Period, you can expect this system, as configured, to deliver one payload in OP2.

5d. Variations in Vessel Spraying Scenarios

Business Rules for Reloading Dispersant Payloads on Vessels

When Resupply on Scene is set to "No", a business rule is in effect that requires a vessel spray platform to return to the staging site when their payload is below 50% of their maximum capacity at the end of an OP. Unlike aircraft, and due to their slow transit speeds, vessels are allowed to reload their payload and transit/return to the spill site between OPs.

In the example Figure below, you can see from the tabular data in the OP1 column, that the vessel platform was able to spray 60% of their payload before OP1 ended. Following the business rule, the vessel then returned to the staging site, reloaded dispersants, and returned to the spill site prior to the start of OP2 (as the one way transit time to and from the spill is only 2 hour each way and vessels are allowed to transit at night between Ops. As a result, the vessel begins OP2 on scene conducting spraying operations and is able to deliver a complete payload before returning back to the staging site.

Estimated Dispersant System Potential (EDSP) (Total Volume of Oil Treated in Operating Period)	Operating Period 1 570 bbl	Operating Period 2 952 bbl

Dispersant Spraying Cycle Timeline

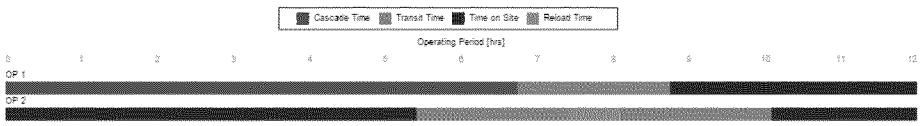
Results for Each Operating Period	OP 1	OP 2
Cascade Time	4.75 hrs	0 hrs
Time On Scene to Commence Spray Operations for OP 1 (Cascade Time + One Way Transit Time)	6.8 hrs	N/A
Adjusted OP Time	7.25 hrs	12 hrs
Payload Deliveries	0.6	1
Dispersant Applied	1196 gal	2000 gal
Total Area Coverage	226 acres	378 acres
Oil Treated (EDSP)	570 bbl	952 bbl
Results per Sortie for a Complete Payload Application		
One-way Transit Time (Including Taxi + Takeoff/Landing for Aircraft)	120 min	
Calculated Pump Rate	4 gpm	
Spray Time/Pass	48 min	
# of Passes/Sortie	10.4	
Spray Time/Sortie	500 min	
Total Time/Sortie	750 min	
Areal Coverage Rate	0.8 acres/min	
Area Covered/Sortie	378 acres	

Figure 55. Business Rule for Reloading Dispersants on Vessels

In the example below, you can see from the tabular data in the OP1 column, that the vessel platform was only able to spray 40% of their payload before OP1 ended due to an increased cascade time from the previous example. Following the business rule, the vessel now remains on scene between OPs and sprays the remaining portion of its payload when OP2 begins. When spraying the full payload is completed, the vessel returns to the staging site to reload dispersants and begin a new sortie.

Estimated Dispersant System Potential (EDSP) (Total Volume of Oil Treated in Operating Period)	Operating Period 1 345 bbl	Operating Period 2 803 bbl

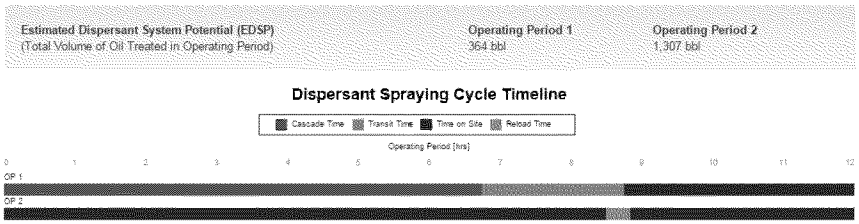
Dispersant Spraying Cycle Timeline



Results for Each Operating Period	OP 1	OP 2
Cascade Time	6.75 hrs	0 hrs
Time On Scene to Commence Spray Operations for OP 1 (Cascade Time + One Way Transit Time)	8.8 hrs	N/A
Adjusted OP Time	5.25 hrs	12 hrs
Payload Deliveries	0.4	0.8
Dispersant Applied	725 gal	1686 gal
Total Area Coverage	137 acres	319 acres
Oil Treated (EDSP)	345 bbl	803 bbl
Results per Sortie for a Complete Payload Application		
One-way Transit Time (Including Taxi + Takeoff/Landing for Aircraft)	120 min	
Calculated Pump Rate	4 gpm	
Spray Time/Pass	48 min	
# of Passes/Sortie	10.4	
Spray Time/Sortie	500 min	
Total Time/Sortie	750 min	
Areal Coverage Rate	0.8 acres/min	
Area Covered/Sortie	378 acres	

Figure 56. Business Rule for Reloading Dispersants on Vessels

When “Resupply On Scene” selection is “Yes”, the vessel platform will be refilled with dispersants whenever the vessel’s dispersant payload has been completely exhausted, and at the end of each OP. When Resupply On Scene selection is “Yes”, vessel platforms that conducted some spray operations during OP 1 will always start out OP 2 with a full payload of dispersants. Setting ‘Resupply On Scene” to Yes negates the need for additional transits between the staging site and the spill for the purposes of reloading dispersants, and increases on scene time for applying dispersants. The Figure below illustrates the change in the same platform’s cycle timeline from the example above if Resupply On Scene is changed from “No” to “Yes”.



Results for Each Operating Period	OP 1	OP 2
Cascade Time	6.75 hrs	0 hrs
Time On Scene to Commence Spray Operations for OP 1 (Cascade Time + One Way Transit Time)	8.8 hrs	N/A
Adjusted OP Time	5.25 hrs	12 hrs
Payload Deliveries	0.4	1.4
Dispersant Applied	765 gal	2745 gal
Total Area Coverage	144 acres	519 acres
Oil Treated (EDSP)	364 bbl	1307 bbl
Results per Sortie for a Complete Payload Application		
One-way Transit Time (Including Taxi + Takeoff/Landing for Aircraft)	120 min	
Calculated Pump Rate	4 gpm	
Spray Time/Pass	48 min	
# of Passes/Sortie	10.4	
Spray Time/Sortie	500 min	
Total Time/Sortie	510 min	
Areal Coverage Rate	0.8 acres/min	
Area Covered/Sortie	378 acres	

Figure 57. Resupply On Scene Set to "Yes"

Multi-Day Cascading of Vessels:

The EDSP output of two operating periods is well suited for aircraft. Cascading of aircraft from out-of-area can be accomplished in most situations in the first operating period so that the estimated dispersant application performance can be displayed for the second operating period. Compared with aircraft, however, vessels are very slow. In order for the EDSP to provide meaningful estimates for vessels, the following logic has been employed based on the Time On Scene to Commence Spray Operations (TOSCO).

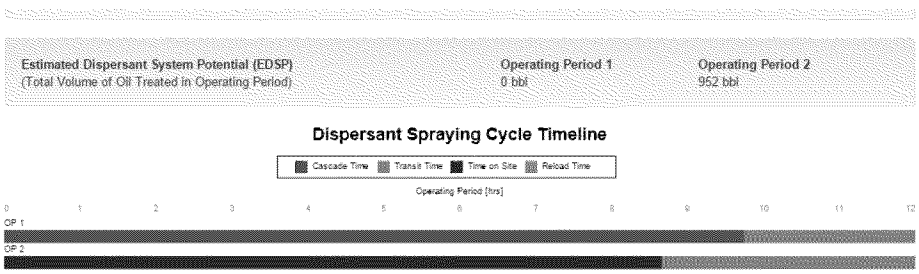
Cascade and Time On Scene Logic for Vessels in the EDSP

- **On Scene During OP1**: If the TOSCO is less than the OP (i.e. the vessel arrives on scene and commences spray operations within OP-1) then show the operations for both OP-1 and OP-2 with the "business rule" (see above) coming into play if OP-1 ends with a partial spray.
- **On Scene In-between OP1 and OP2:** If the TOSCO is greater than OP time (past OP-1) but Less than 24 hours, (i.e. the vessel arrives on scene ready to spray after the end of OP-1 but before the start of OP-2) then the OP-1 timeline

reflects any actual Cascade and Transit that occurred during the first OP, and the second OP starts with a spray operation.

- **On Scene During OP2**: If the TOSCO is greater than OP time (arrives after OP-1) but Less than 24 + OP-hours, then the first and second operating periods reflect the actual Cascade and transit times, and the vessel arrives and starts spray operations during the second operating period.
- **On Scene After the end of OP2**: If TOSCO is greater than OP hours + 24 hours (multiday cascade), then the first operating period will begin with a spray operation. The labels for the timelines and the tabular data columns will be changed from "OP-1" and "OP-2" to "1st Full OP On Scene" and "2nd OP On Scene". Since this scenario will lead to starting OP-1 with a spray operation, the other output fields in the first column headed "1st Full OP On Scene", will appear as they normally would for an operating period starting with a spray operation.

In the example below, the vessel had a cascade time of 9.75 hours and a time on scene of 15.8 hours. Since the vessel arrived in between OP1 and OP2, the vessel timeline will start out with spraying operations at the beginning of OP2.



Results for Each Operating Period	OP 1	OP 2
Cascade Time	9.75 hrs	0 hrs
Time On Scene to Commence Spray Operations for OP 1 (Cascade Time + One Way Transit Time)	15.8 hrs	N/A
Adjusted OP Time	2.25 hrs	12 hrs
Payload Deliveries	0	1
Dispersant Applied	0 gal	2000 gal
Total Area Coverage	0 acres	378 acres
Oil Treated (EDSP)	0 bbl	952 bbl
Results per Sortie for a Complete Payload Application		
One-way Transit Time (Including Taxi + Takeoff/Landing for Aircraft)	360 min	
Calculated Pump Rate	4 gpm	
Spray Time/Pass	48 min	
# of Passes/Sortie	10.4	
Spray Time/Sortie	500 min	
Total Time/Sortie	1230 min	
Areal Coverage Rate	0.8 acres/min	
Area Covered/Sortie	378 acres	

Figure 58. Vessel Arriving On Scene in between OP1 and OP2

The Figures below illustrate a vessel cascading over multiple OPs and arriving on scene after the end of OP2. In this case, a vessel of opportunity required 24 hours for mobilization before transiting 50 miles to the staging site to load dispersants. The cascade time was 34.75 hours, and the time on scene was 42.8 hours. Since the vessel arrived after the end of OP2, the outputs will show the first two full OPs where the vessel is on scene. In this case, that would be for OP3 and OP4, as OP3 starts at 48 hours.

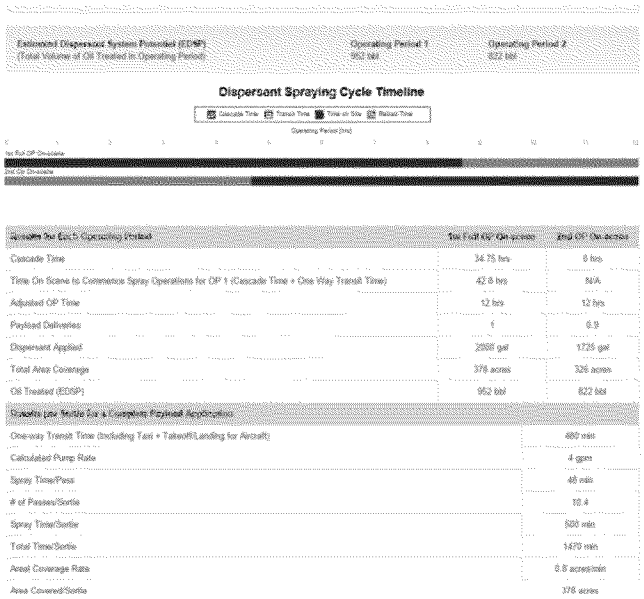


Figure 59. Vessel of Opportunity Arriving On Scene after OP2

If resupply on scene is then set to “Yes”, the outputs change as seen below. With resupply set to “Yes”, the output will repeat itself for each full OP.

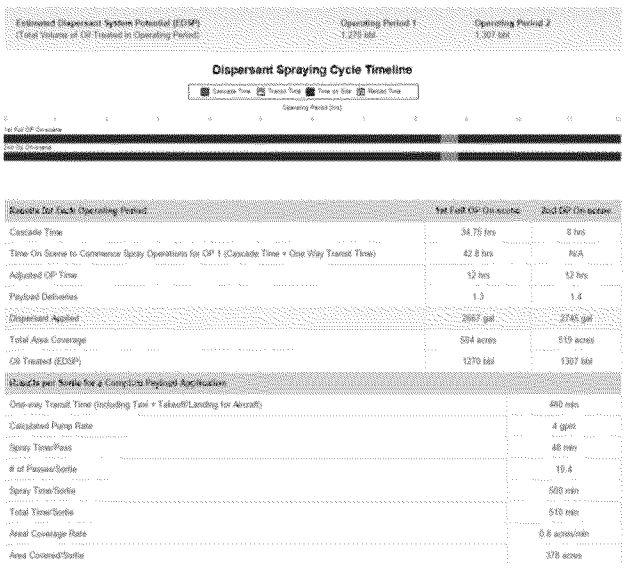


Figure 60. Vessel of Opportunity Arriving On Scene after OP2 with Resupply On Scene

6. Equations Used in Calculations

In the EDSP Calculator, the following variables and equations are used to estimate rates, times, areal coverage, and volumes.

Cascade Time

If the Cascade Distance to Staging Site is less than or equal to the Maximum Flight Range then the cascade can be done with one leg and,

Aircraft Cascade Time = Mobilization Time + (Taxi + Takeoff/Landing Time) + Distance to Staging Site/Cascade Transit Speed + (Taxi + Takeoff/Landing Time) + 45 minute briefing period.

The Cascade Transit Speed and Maximum Flight Range for aircraft will vary depending on whether the platform is cascading with or without a dispersant payload.

If the Cascade Distance is greater than the Maximum Flight Range then the cascade will require multiple legs. Each leg will require a Fuel Load Time.

Vessel Cascade Time = Mobilization Time + Distance to Staging Site/Cascade Transit Speed + 45 minute briefing period.

One-way Transit Time

Aircraft One-way Transit Time = (Taxi + Takeoff/Landing Time) + (One-way Transit Distance, staging to & from spill) /Transit Speed

Vessel One-way Transit Time = (One-way Transit Distance, staging to & from spill) /Transit Speed

Areal Coverage Rate

Areal Coverage Rate = Application Speed x Swath Width/430

Pump Rate

Pump Rate = Dosage x Areal Coverage Rate

Spray Time/Sortie

Spray Time/Sortie = Payload/Pump Rate

Spray Time/Pass

Spray Time/Pass = Average Spray Pass Length/Application Speed

of Passes/Sortie

# of Passes/Sortie = (Spray Time/Sortie) / (Spray Time/Pass)

The integer part of this is the number of complete passes. The decimal part of this determines the extent of the last partial pass before the end of the sortie.

For Bidirectional Spraying:

Number of U Turns = INT(# of Passes/Sortie)

Number of Approaches + Departures = 2 x INT(# of Passes/Sortie) + 2

For Unidirectional Spraying:

Return Leg Distance = Approach + Average Spray Pass Length + Departure

Return Leg Time = Return Leg Distance / Reposition Speed

Number of U Turns = 2 x INT(# of Passes/Sortie)

Number of Approaches + Departures = 4 x INT(# of Passes/Sortie) + 2

Maximum Transit Distance

The Maximum Transit Distance is the maximum distance for a particular scenario and aircraft platform combination that the platform can reach, deliver a full payload, and return to the staging site within its Max Sortie Time.

Time On Station = 10 min check in time + Approaches/Departures + spray passes + U Turns.

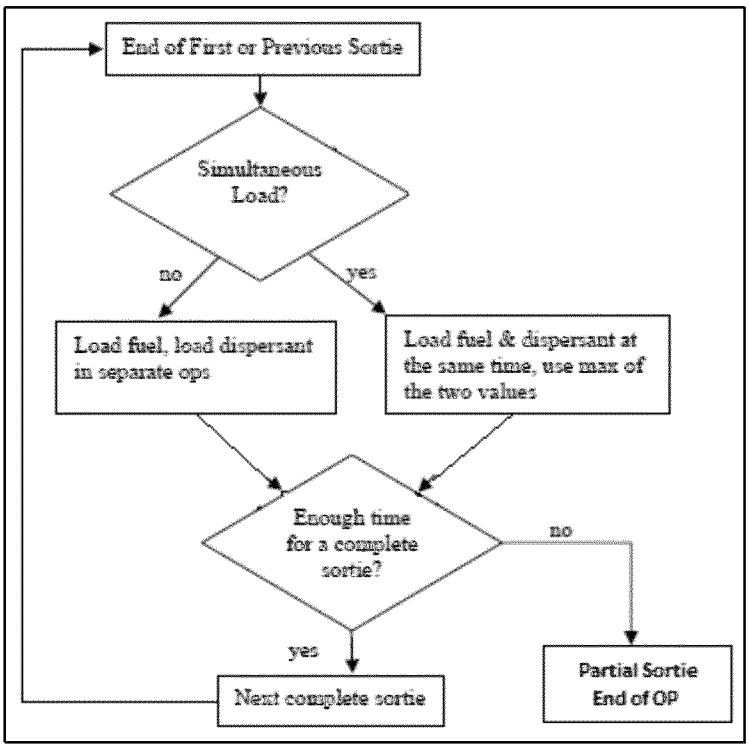
Total Time/Sortie = Time On Station + 2(One-Way Transit Time)

Maximum One-way Transit Time = (Max Sortie Time – Time On Station) / 2

Maximum Transit Distance = Maximum One-way Transit Time x Transit Speed

Payload Deliveries

The following logic is used to determine the Payload Deliveries in an OP (EDAC) and Oil Treated (EDSP):



Cycle Time = Total Time/Sortie + Resupply Time

Payload Deliveries = INT(Operating Period Time / Cycle Time)

Dispersant Applied = Payload Deliveries x Payload

This will be the number of complete payload deliveries in the operating period. Next add any partial sortie spray passes that can be accomplished in the remaining time in the operating period.

OP Time Remaining = Operating Period Time – (Payload Deliveries x Cycle Time) – Transit Time – Check in Time – Approach Time

If OP Time Remaining > 0

Then if OP Time Remaining > Spray Time/Pass

Dispersant Applied = Dispersant Applied + (Spray Time/Pass x Pump Rate)

OP Time Remaining = OP Time Remaining – Departure Time – U Turn Time – Approach Time (for bidirectional spraying)

OR

OP Time Remaining = OP Time Remaining – Departure Time – U Turn Time – Return Leg Time – U Turn Time (for unidirectional spraying)

End if

End if

Now add spray time for final partial pass

Dispersant Applied (EDAC) = Dispersant Applied + OP Time Remaining x Pump Rate

EDSP = Dispersant Applied x DOR/42

Total Spray Time in OP = Dispersant Applied / Pump Rate

Total Area Coverage = Areal Coverage Rate x Total Spray Time in OP

To: Wilson, Gregory[Wilson.Gregory@epa.gov]
From: Principe, Vanessa
Sent: Wed 1/25/2017 7:21:41 PM
Subject: FW: State-of-Science: Eco-toxicity and Sublethal Impacts final document review
mace edits to 2017.01.09_EcoToxicity and Sublethal Impacts.docx

From: Barron, Mace
Sent: Saturday, January 21, 2017 1:45 PM
To: Mandsager, Kathy <kathy.mandsager@unh.edu>; sarah.allan@noaa.gov; abejarano@researchplanning.com; DFrenchMcCay@asascience.com; michel.gielazyn@noaa.gov; peter.hodson@queensu.ca; sharon.hook@csiro.au; rrhopcroft@alaska.edu; john.incardona@noaa.gov; Angela_Matz@fws.gov; Teri.Rowles@noaa.gov; Mathijs.Smit@shell.com; Sprenger, Mark <Sprenger.Mark@epa.gov>; rstjeerdema@ucdavis.edu; dana@mote.org; john.wise@louisville.edu; mhziccardi@ucdavis.edu
Cc: nancy.kinner@unh.edu; Gary Shigenaka - NOAA Federal <gary.shigenaka@noaa.gov>; doug.helton@noaa.gov; Principe, Vanessa <Principe.Vanessa@epa.gov>; Wilson, Gregory <Wilson.Gregory@epa.gov>
Subject: RE: State-of-Science: Eco-toxicity and Sublethal Impacts final document review

Ok, just a few comments in the attached from me.

Wondering if we can move this to manuscript phase soon?

From: Mandsager, Kathy [mailto:kathy.mandsager@unh.edu]
Sent: Friday, January 13, 2017 8:14 AM
To: sarah.allan@noaa.gov; Barron, Mace <Barron.Mace@epa.gov>; abejarano@researchplanning.com; DFrenchMcCay@asascience.com; michel.gielazyn@noaa.gov; peter.hodson@queensu.ca; sharon.hook@csiro.au; rrhopcroft@alaska.edu; john.incardona@noaa.gov; Angela_Matz@fws.gov; Teri.Rowles@noaa.gov; Mathijs.Smit@shell.com; Sprenger, Mark <Sprenger.Mark@epa.gov>; rstjeerdema@ucdavis.edu; dana@mote.org; john.wise@louisville.edu; mhziccardi@ucdavis.edu
Cc: nancy.kinner@unh.edu; Gary Shigenaka - NOAA Federal <gary.shigenaka@noaa.gov>; doug.helton@noaa.gov; Principe, Vanessa <Principe.Vanessa@epa.gov>; Wilson, Gregory <Wilson.Gregory@epa.gov>; Mandsager, Kathy <kathy.mandsager@unh.edu>
Subject: State-of-Science: Eco-toxicity and Sublethal Impacts final document review

Dear Scientific Panel,

We would very much appreciate your review of this final version of the *Eco-toxicity and Sublethal Impacts* document. This document incorporates the edits and suggestions made via email and final discussion.

If you could reply by email with your approval of this document by **Friday, January 27** (2 weeks for this review) that would be greatly appreciated. I will need each of you that participated in this discussion to reply in the affirmative that you approve this document. [The next step is for this document to be submitted publicly for input.]

Please also confirm that your name and affiliation are listed correctly at the end of this document.

You deserve kudos for this great accomplishment! Thank you very much!

Kathy Mandsager

Coastal Response Research Center

Center for Spills and Environmental Hazards

220 Gregg Hall, 35 Colovos Rd

University of New Hampshire

Durham, NH 03824

603.862.1545

To: Wilson, Gregory[Wilson.Gregory@epa.gov]
From: Principe, Vanessa
Sent: Fri 7/31/2015 6:04:10 PM
Subject: RE: REMINDER - ICCOPR R&T PLAN REVIEW
ICCOPR RT Plan Draft for Members 23JULY2015 v1 line nmbr_OEM_7_31_15.docx
Copy of RT Plan Comment Matrix 23JULY2015_OEM_7_31_15.xls

Bill:

Attached are our comments on the revised R&T Plan. We are sending you both a populated spreadsheet and an annotated word document as a backup. Most of our concerns are in the newly developed Definitions section, and on characterization of dispersants use.

We want to also request you allow for Robyn, as the official EPA ICCOPR member, to have the opportunity to review and add any comments; we understand she is out of the office until August 7, so she may need a bit of extra time beyond your due date. Further, for those comments we have indicated are critical to EPA, we would like an opportunity to review the changes made to reflect those concerns prior to bringing the Plan to the ICCOPR membership for a final vote.

FYI, both Vanessa and I will be on leave until the week of August 24, and will look for any revised version addressing our comments first thing as we get back.

We thank you again for the opportunity to review and for working to address our concerns.

-----Original Message-----
From: Vocke, William CIV [mailto:William.T.Vocke@uscg.mil]
Sent: Thursday, July 30, 2015 3:30 PM
To: Wilson, Gregory
Cc: Principe, Vanessa; Conmy, Robyn
Subject: RE: REMINDER - ICCOPR R&T PLAN REVIEW

Thanks Greg.

I know several members are on/going on leave. But we've all had a good chance to review portions. So please send any comments you have.

Hope you and Vanessa both have good vacations.

Bill

-----Original Message-----
From: Wilson, Gregory [mailto:Wilson.Gregory@epa.gov]
Sent: Thursday, July 30, 2015 3:22 PM
To: Vocke, William CIV
Cc: Principe, Vanessa; Conmy, Robyn
Subject: RE: REMINDER - ICCOPR R&T PLAN REVIEW

Thanks for the reminder. Just an fyi, I believe Robyn is out until 8/7, and Vanessa and I are going on annual leave after tomorrow. We are trying to get our review done now, but wanted to alert you that folks are out of pocket.

-----Original Message-----
From: Vocke, William CIV [mailto:William.T.Vocke@uscg.mil]
Sent: Thursday, July 30, 2015 2:48 PM

To: Abe.nachabe@navy.mil; Andrew.J.Bruzewicz@usace.army.mil; Balsley, Alexander CIV; barry_Forsythe@fws.gov; Bowis, Meagan K LT; Carnegie, Tammie R LT; christina_kravitz@fws.gov; Conmy, Robyn; Crecy, Stacey L LCDR; daniel.yuska@dot.gov; dave.westerholm@noaa.gov; David.Lehman@dot.gov; david.moore@bsee.gov; debbie.payton@noaa.gov; DiRenzo, Joseph CIV; Elena.Melchert@hq.doe.gov; Erica.Folio@Hq.Doe.Gov; Fletcher, James E CIV; frank.stone@navy.mil; Hall, Gregory CDR (EDU); kurt.a.hansen@uscg.mil; holly_herod@fws.gov; jason.boehm@nist.gov; Jeff.Ji@boem.gov; Jenkins, Shannon R CIV; jfarrell@arctic.gov; Jim Merritt; john.kucklick@nist.gov; john.kucklick@noaa.gov; Kemp.skudin@navy.mil; kevin.easley@hq.doe.gov; Lori.Medley@bsee.gov; Loring, Joseph B CAPT; Lundgren, Scott; mfocazio@usgs.gov; michael.carter@dot.gov; michael.j.green-1@nasa.gov; Monica Maghini; Platt, Jeffrey R LT; Principe, Vanessa; rachel.hardey.ctr@navy.mil; robb.hyde@navy.mil; robert.w.smith@dot.gov; Scott Lundgren NOAA; Stephanie A. Brown; steve.lehmann@noaa.gov; steven.pearson@bsee.gov; Steven.V.Cary@usace.army.mil; Turner, Arden C CIV; mark.g.vanhaverbeke@uscg.mil; Vocke, William CIV; Walter.Johnson@boem.gov; wayne.yoder@dhs.gov; Weaver, James (USCG); Wilson, Gregory
Subject: REMINDER - ICCOPR R&T PLAN REVIEW

All,

This is just a friendly reminder that comments on the Draft ICCOPR R&T Plan are due to me by COB August 7, 2015. Please take some time and review the plan. This is the final round of edits before the Plan is put up for a vote.

Bill

To: Wilson, Gregory[Wilson.Gregory@epa.gov]
From: Principe, Vanessa
Sent: Tue 11/24/2015 8:39:15 PM
Subject: Maybe this is what you remember?

SEE YELLOW HIGHLIGHT BELOW - This is the other place we've come across this thought recently.

The Poisoning / Rolling Stone Article

http://www.rollingstone.com/politics/news/the-poisoning-20100721?page=5

From: Principe, Vanessa
Sent: Tuesday, November 17, 2015 1:24 PM
To: Barron, Mace <Barron.Mace@epa.gov>
Cc: Wilson, Gregory <Wilson.Gregory@epa.gov>
Subject: RE: comments on fact sheet....

Mace:

Here are some observations/recommendations. Most important is to let them know that this is in no way and EPA official review, just the opinion of the individual reviewers. For your responses below, we highlighted the ones for which we have a different take/suggestion. Take a look to see if you agree. Let us know if you want to chat.

Feel better, G & V

Overall observation – In trying to present "both sides of the story" perhaps the document would benefit from a balanced approach to presenting the information similar to what is described in NRC – reference (7) at page 2: "Oil spill dispersants do not actually reduce the total amount of

oil entering the environment. Rather, they change the inherent chemical and physical properties of oil, thereby changing the oil's transport, fate, and potential effects. Small amounts of spilled oil naturally disperse into the water column, through the action of waves and other environmental processes.

The objective of dispersant use is to enhance the amount of oil that physically mixes into the water column, reducing the potential that a surface slick will contaminate shoreline habitats or come into contact with birds, marine mammals, or other organisms that exist on the water surface or shoreline. Conversely, by promoting dispersion of oil into the water column, dispersants increase the potential exposure of water-column and benthic biota to spilled oil. Dispersant application thus represents a conscious decision to increase the hydrocarbon load (resulting from a spill) on one component of the ecosystem (e.g., the water column) while reducing the load on another (e.g., coastal wetland). Decisions to use dispersants, therefore, involve trade-offs between decreasing the risk to water surface and shoreline habitats while increasing the potential risk to organisms in the water column and on the seafloor. This trade-off reflects the complex interplay of many variables, including the type of oil spilled, the volume of the spill, sea state and weather, water depth, degree of turbulence (thus mixing and dilution of the oil), and relative abundance and life stages of resident organisms."

Overall Observation – Check the links throughout the document, many seem to be dead.

In the document:

The Corexit products that were used in DWH were Corexit 9500A and 9527A. Corexit is a trademarked name; there are different formulations and product types under that trade name listed in the NCP Product Schedule, including non-dispersants. Furthermore, the NCP Product Schedule now also includes a Corexit 9500B dispersant product. Should specify 9527A and 9500A throughout as the specific Corexit products used in DWH, and should not include "corexit" in the glossary solely as a dispersant without the specific number designation.

"**Regulations state** that responders spraying dispersant from airplanes could not apply them within 2.3 miles of any vessel, within 3.45 miles of the shoreline or visible marine life, or in areas with water depths less than 33 feet deep." – These are operational parameters that were specific to the DWH response. While some of them may have been specified in pre-authorization plans, they are not coded in regulations.

"This method reduced the amount of dispersant in the air, increasing the safety of the working environment." – Please cite source for this statement. If this statement was intended to be tied with the subsequent statement, then please clearly link them. Suggest: "According to the U.S. Coast Guard, this method reduced the amount of dispersant in the air, increasing the safety of

the working environment. The same Coast Guard report states that emergency responders applied 98 percent of the dispersants more than 11 miles offshore."

"The smaller droplets mix into the water column and then are carried away by currents, settle on the bottom, or evaporate." - Biodegradation is not the sole fate/transport mechanism that is at play after dispersant use. Furthermore, there are certain oil components that are recalcitrant to biodegradation. According to the NRC reference (7): "These small submerged oil droplets are then subject to transport by subsurface currents and other natural removal processes, such as dissolution, volatilization from the water surface, biodegradation, and sedimentation resulting from interactions with suspended particulate material (SPM)." (pg 51) Figure 4-1 provide a visual representation of these major open-ocean oil fate and transport processes. (pg 136)

"Emergency responders typically use dispersants when other cleanup methods will not be effective." – During DWH more than one response technology was used concurrently. The use of one is not exclusive of the other. Suggest deleting sentence and starting the following sentence with "Emergency responders may use dispersants when, for example…"

"Dispersants can prevent an oil slick from reaching the shoreline." – This rationale is not unique to dispersants. Furthermore, while protecting the shoreline may be a priority, depending on the habitats present at the spill site, protection of the water column organisms may also play an important role in dispersant use decisions.

"Agencies in charge of the response efforts approved the deep-water application of dispersants because it minimized human contact with dispersants. This technique also allowed for a quicker response time because dispersants could be applied at night and in foul weather." – Citation within the referenced Commission Staff Working Paper No. 4 (38) doesn't seem to support the statement. – "Jeff Goodell, The Poisoning, ROLLING STONE (July 21, 2010). Administrator Jackson gave a wide-ranging and candid interview to Rolling Stone, in which she stated that she had told her aides that the approval decision was among the hardest she had ever made. She also reportedly said that BP argued for subsea application as a method that would reduce the overall volume of chemicals discharged into the marine ecosystem. Id." Perhaps the attempt to paraphrase changed the meaning. Consider if a direct quote from the Paper No. 4 may be a better approach: "Considerations related to response worker health and ease of application—subsea application would minimize the necessary human contact with dispersants, and could occur at night and in foul weather—reportedly played a role in the decision to approve the method."

The cited Commission Staff Working Paper No. 4 is not available through the link (reference 1).

The version found at http://cybercemetery.unt.edu/archive/oilspill/20121211010455/http://www.oilspillcommission.gov/sites/defaul has the following disclaimer: "Staff working papers are written by the staff of the National Commission on the BP Deepwater Horizon Oil Spill and Offshore Drilling for the use of members of the Commission. They do not necessarily reflect the views of the Commission or any of its members. In addition, they may be based in part on confidential interviews with government and non-government personnel." At a glance, it gives the impression that what is being cited is the Commission report.

"The National Contingency Plan (NCP) guidelines are used to regulate dispersant use. These guidelines outline which dispersants emergency responders can apply during a spill, where they can apply them, and how much they can apply." – The NCP are regulatory requirements, not guidelines. Further the NCP regulations establish the framework for emergency responders to authorize the use of dispersants during a response, including considerations of where and how much.

"Emergency responders use Special Monitoring of Applied Response Technologies (SMART) guidelines to monitor dispersant use." – SMART protocols are not applicable to subsea. Clarify they are "…to monitor surface dispersant use."

"The U.S. Coast Guard, National Oceanic and Atmospheric Administration (NOAA), U.S. Environmental Protection Agency (EPA), U.S. Centers for Disease Control and Prevention, and the U.S. Bureau of Safety and Environmental Enforcement are all part of the monitoring program." – As in these entities authored the SMART Protocols, or as in they all participated in monitoring during the DWH event? Unclear.

"The U.S. EPA monitored the air quality and tested for two chemicals, 2-butoxyethanol and propylene glycol. Both of these chemicals are found in Corexit." - Table 2 below shows components as an aggregate of all chemicals that can be present. However, it further specifies that 2-Butoxyethanol is "Found in Corexit 9527A only, not Corexit 9500". The statement as written is inconsistent with Table 2.

"Scientists took samples 30 minutes after dispersant application and did not find either chemical." – Reference cited is dead link to the SMART Protocols, which would not be relevant to this statement. Where was this information?

Table 1 – refers to toxicity in general, where it should speak to acute toxicity, as those are the levels for aquatic toxicity cited. Also, the discussion surrounding the table goes to air

monitoring.

"The Gulf of Mexico Research Initiative (GoMRI) and other research programs are looking at alternative dispersant use and technology, how dispersants react with oil under various conditions, the environmental impacts of currently approved dispersants, and the impacts that dispersants have on human health." – Dispersants, nor any other agent on the NCP Product Schedule, is "approved." The NCP listing of product on the Schedule does not constitute approval or endorsement of that product, nor a recommendation of its use. To avoid any possible misinterpretation or misrepresentation

of this policy the Agency requires a disclaimer to be included on any label, advertisement, or technical literature for the product. Please refer to "dispersants listed on the NCP Product Schedule."

Table 2 – Cite source for column 2.

Glossary –

Corexit is a trade name that includes products other than dispersants. Need to refer to specific product number. Would remove "modern" qualifier." Again, Products on the NCP product Schedule are not "approved." Please refer to the specific Corexit dispersant products as "listed on the NCP Product Schedule." http://www2.epa.gov/emergency-response/ncp-product-schedule-products-available-use-oil-spills

Dispersants - From the NRC reference (7) at page 51: "Dispersants are mixtures of solvents, surfactants, and other additives that are applied to oil slicks to… promotes the formation of a larger number of small oil droplets when surface waves entrain oil into the water column. These small submerged oil droplets are then subject to transport by subsurface currents and other natural removal processes, such as dissolution, volatilization from the water surface, biodegradation, and sedimentation resulting from interactions with suspended particulate material (SPM)."

NCP - The National Oil and Hazardous Substances Pollution Contingency Plan, more commonly called the National Contingency Plan or NCP, is the federal government's regulatory blueprint for responding to both oil spills and hazardous substance releases.

SMART - *Special Monitoring of Applied Response Technologies* is a cooperatively designed

monitoring program for in situ burning and surface dispersants. SMART relies on small, highly mobile teams that collect real-time data using portable, rugged, and easy-to-use instruments during surface dispersant and in situ burning operations.

From: Wilson, Gregory
Sent: Monday, November 16, 2015 11:23 AM
To: Principe, Vanessa <Principe.Vanessa@epa.gov>
Subject: FW: comments on fact sheet....

From: Barron, Mace
Sent: Thursday, November 12, 2015 5:53 PM
To: Larissa Graham <ljg0008@auburn.edu>
Cc: Wilson, Gregory <Wilson.Gregory@epa.gov>
Subject: comments on fact sheet....

Greg may have additional comments. I forwarded the document to him.

Thanks for allowing me to review. As discussed previously, my review is not an official EPA review.

Mace

1) check the volume of oil; the court decided on a final number that I think is less than you list.

2) reword this:

"Both are on the federal guideline list of dispersants approved for use during a spill."

*the list is not approved: that is done by the incident command. Alternative wording:

"Both dispersants are listed in the U.S. National Contingency plan."

3) double check this statement: "dispersants over 300 square miles of oiled"

*not sure how the surface area of slicks that were sprayed would have been determined. Surface area sprayed does not equal surface area of oil sprayed, as that would assume 100% hitting the target.

4) If want to be balanced in section on how dispersants work, would including something like:

"Dispersants also increase the amount of petroleum hydrocarbons into the water column, which can increase toxicity to aquatic organisms."

5) I would reword this statement:

"when other cleanup methods will not be effective" because it is rarely true. Often dispersants are used because they are easier than mechanical recovery.

*rather: "when other cleanup methods may not be as expedient or practical"

6) reword this:

"because it minimized human contact with dispersants."

to:

"because it reduced the volume of oil reaching the ocean surface and minimized human contact with dispersants."

7) reword this:

"This technique also allowed for a quicker response time because dispersants could be applied at night and in foul weather."

to:

"This technique may have been more effective at dispersing the oil because of direct injection into the newly released oil, and also allowed for a quicker response time because dispersants could be applied at night and in foul weather."

8) Reword this sentence because the NCP does not specify where, how much. That is upto the incident command.

"These guidelines outline which dispersants emergency responders can apply during a spill, where they can apply them, and how much they can apply"

to:

"These guidelines outline which dispersants emergency responders can apply during a spill, and the monitoring requirements during dispersant use".[9]

9) I was not aware of the air sampling for the corexit components, and did not try to confirm your info.

10) I think showing the LC50 stuff in Table 1 is confusing. The information in and around this table is human health. Not sure of the value of having ecotox, especially when the species are not even the same.

11) This statement is an overstatement because we only tested acute toxicity, not reproduction etc; reword:

"when tested alone, caused serious impacts to the shrimp or silversides.

to:

"when tested alone were less toxic to the shrimp or silversides than oil."

12) reword because not specifically correct:

"was no more or less toxic than seven other dispersants that were tested"

to:

"had similar toxicity as the seven other dispersants that were tested"

13) the observation photo seems to show that spraying occurred where there was no oil. I think that happened a lot (ie, they turned on the spray before they were over the oil; slicks were dispersed already, rather than a continuous slick. I think the 300 miles cited above is likely inaccurate...but I understand you need to just cite publically available info.

14) I would delete 'air freshener" from table 2. Seems unlikely that the petroleum mixture in corexit is the same.

15) reword this definition on dispersants because it implies 100% efficacy:

"prevent floating oil from impacting sensitive ecosystems such as coastal habitats."

to:

"can limit floating oil from impacting sensitive ecosystems such as coastal habitats."

From: Larissa Graham <ljg0008@auburn.edu>
Sent: Thursday, November 12, 2015 3:33 PM
To: Barron, Mace
Subject: RE: Your help needed - review?

Hi Mace,

Just wanted to check in regarding our publication on dispersants. I'm particularly interesting in your feedback on Tables 1 & 2. We've been asked what chemicals are found in dispersants and were trying to present information in a way that would tell both sides of the story.

Please let me know if you have any questions.

Looking forward to your thoughts!

Thanks,

Larissa

Larissa Graham

Oil Spill Science Extension Specialist

Mississippi-Alabama Sea Grant Consortium

Office: (251) 438-5690

Cell: (251) 348-5436

Larissa.Graham@auburn.edu

From: Larissa Graham
Sent: Friday, November 6, 2015 10:32 AM
To: 'Barron, Mace' <Barron.Mace@epa.gov>
Subject: RE: Your help needed - review?

Great, thank you!

Have a nice weekend,

Larissa

Larissa Graham

Oil Spill Science Extension Specialist

Mississippi-Alabama Sea Grant Consortium

Office: (251) 438-5690

Cell: (251) 348-5436

Larissa.Graham@auburn.edu

From: Barron, Mace [mailto:Barron.Mace@epa.gov]
Sent: Friday, November 6, 2015 10:24 AM
To: Larissa Graham <ljg0008@auburn.edu>
Subject: Re: Your help needed - review?

I will get you comments next week Larissa.

From: Larissa Graham <ljg0008@auburn.edu>
Sent: Thursday, November 5, 2015 3:08 PM
To: Raimondo, Sandy; Barron, Mace
Cc: Wilson, Gregory; Steve Sempier

Subject: RE: Your help needed - review?

Hi Sandy,

Ok, thank you for letting us know.

Mace,

If you were willing to review as a scientist, we would really appreciate it.

Thanks,
Larissa

Larissa Graham

Oil Spill Science Extension Specialist

Mississippi-Alabama Sea Grant Consortium

Office: (251) 438-5690

Cell: (251) 348-5436

Larissa.Graham@auburn.edu

From: Raimondo, Sandy [mailto:Raimondo.Sandy@epa.gov]
Sent: Thursday, November 5, 2015 2:51 PM
To: Larissa Graham <ljg0008@auburn.edu>
Cc: Barron, Mace <Barron.Mace@epa.gov>; Wilson, Gregory <Wilson.Gregory@epa.gov>; Steve Sempier <stephen.sempier@usm.edu>
Subject: Re: Your help needed - review?

Mace, Larrissa,

Mace can provide a peer review as a scientist, but cannot provide an official EPA perspective.

Thanks,

Sandy

On Nov 5, 2015, at 1:46 PM, Larissa Graham <ljg0008@auburn.edu> wrote:

Hi Mace,

Sounds good. If we need you to review as a scientist, that's ok too. We just really want to make sure our information is accurate.

Please let me know what you find out.
Thanks,

Larissa

Larissa Graham

Oil Spill Science Extension Specialist

Mississippi-Alabama Sea Grant Consortium

Office: (251) 438-5690

Cell: (251) 348-5436

Larissa.Graham@auburn.edu

From: Barron, Mace [mailto:Barron.Mace@epa.gov]
Sent: Thursday, November 5, 2015 2:26 PM
To: Larissa Graham <ljg0008@auburn.edu>
Cc: Wilson, Gregory <Wilson.Gregory@epa.gov>; Raimondo, Sandy <Raimondo.Sandy@epa.gov>
Subject: Re: Your help needed - review?

Am happy to review. However, I ccd my branch chief Sandy and an Office of Emergency Management expert, Greg, as an fyi to see if there are any issues with what may be perceived as an Agency official review.

For example, there are likely no issues with me providing you a technical review as a scientist...but am uncertain about inferring EPA review aspects.

I will wait for Greg and Sandy to provide their thoughts before I open up the doc.

Mace

From: Larissa Graham <ljg0008@auburn.edu>
Sent: Thursday, November 5, 2015 1:07 PM
To: Barron, Mace
Subject: Your help needed - review?

Hi Mace,

I heard you met Emily! I'm glad you two were able to meet up and chat.

I'm looking forward to our meeting on the 17th but I was wondering if you could help me with something before then. I've been working on an outreach publication about dispersants. It's the first in our dispersant series and mainly provides some background information. Some of our reviewers wanted us to include information about the chemicals found in Corexit and Corexit, itself, so I added Tables 1 & 2.

Would you be willing to look over this publication on behalf of EPA and let me know what you think about the tables? If you have any other comments, I would love to hear them. We want to make sure we are sharing the best information possible, in a non-biased way.

Thank you!

Larissa

Larissa Graham

Oil Spill Science Extension Specialist

Mississippi-Alabama Sea Grant Consortium

Office: (251) 438-5690

Cell: (251) 348-5436

Larissa.Graham@auburn.edu

April 22, 2015

U.S. Environmental Protection Agency
EPA Docket Center (EPA/DC)
Docket ID No. EPA–HQ–OPA–2006–0090
1200 Pennsylvania Avenue NW 20460

Submitted via electronic submission to http://www.regulations.gov **and by mail**

Dear Dr. Gregory Wilson,

On January 22, 2015, the U.S. Environmental Protection Agency (EPA or the Agency) issued a Proposed Rule to amend Subpart J of the National Oil and Hazardous Substances Contingency Plan that governs the use of dispersants, other chemical and biological agents, and other spill mitigating substances when responding to oil discharges into the waters of the United States. 80 Fed. Reg. 3380 (January 22, 2015). Comments are due by April 22, 2015. The following comments to the docket are provided by the American Petroleum Institute (API) and the National Ocean Industries Association (NOIA) (collectively, "we") on behalf of our member companies based on review of the above-cited Proposed Rule.

API is a national trade association that represents over 625 members involved in all aspects of the oil and natural gas industry, including the exploration and production of both onshore and offshore Federal resources. The U.S. oil and natural gas industry supports 9.8 million U.S. jobs and 8 percent of the U.S. economy, delivering tens of millions per day in revenue to our government. API members provide most of the nation's energy and are backed by a growing grassroots movement of more than 25 million Americans.

NOIA, founded in 1972, represents more than 270 companies among all segments of the offshore industry with an interest in the exploration and production of both traditional and renewable energy resources on the nation's outer continental shelf (OCS). NOIA's mission is to secure reliable access and a fair regulatory and economic environment for the companies that develop the nation's valuable offshore energy resources in an environmentally responsible manner.

API, NOIA, and our respective members recognize the important mission of the EPA in protecting human health and the environment and share in this mission. More than most industries, the oil and natural gas industry understands the nature of risk and places at the center of its concern comprehensive planning for and effective management of risks in order to safeguard human well-being and protect environmental resources. To this end, woven into every operational endeavor are release prevention measures, preparedness methods and plans for rapid, effective response should an incident resulting in a release occur. Developed alongside regulatory agencies, operators submit to and follow a distinct set of response principles which protect shared values. The framework by which these principles are employed and tradeoffs are made in order to protect these shared values is called Net Environmental Benefit Analysis (NEBA). NEBA is a process used by the response community for making the best choices to minimize impacts of oil releases on people and the environment. NEBA aids in protecting people and the environment, and is the matrix through which every decision is made in a response event. In the offshore context, oil dispersing agents are a primary tool in the NEBA process and the response toolkit, as they

are a principal tool used for protecting human health and defending environmentally sensitive areas. When a release occurs, judgment and response speed are critical. In an offshore setting, dispersants are essential in creating a safe environment and are a key ingredient in marshaling an effective, swift response.

API, NOIA, and our respective members understand that through this Proposed Rule, EPA's goal is to develop a regulation which protects people and preserves natural resources. We too wish to accomplish the same goal, based upon sound scientific research and grounded in real-world, outside-the-laboratory scenarios and taking into account incident-specific risk. While the original intent of the dispersant rule was simply to provide for a schedule of dispersants and other chemicals that may be used in carrying out the National Contingency Plan (NCP), the Proposed Rule would essentially expand its purpose to an operational rulemaking which would significantly alter operational procedures.

In summary, the Proposed Rule, if implemented, would be substantially problematic for all stakeholders, including the response community, which is required to carry out an effective spill response. The Proposed Rule would also be problematic for the Federal and state ag ency community interests, as well as the public with regard to increasing environmental and community impacts. Although the Agency indicates that it seeks to "*encourage the development of safer and more effective spill mitigating products, and would bette r target the use of these products to reduce the risks to human health and the environment*" and "*ensure that On-Scene Coordinators (OSCs), Regional Response Teams (RRTs), and Area Committees have sufficient information to support agent preauthorization or authorization of use decisions*," API, NOIA, and our respective members submit that the Proposed Rule falls short of these objectives and in many cases would have the opposite effect . For example, and at the center of our concern, is the proposed dispersed oil toxicity test and its threshold which could result in the elimination of many dispersants and potential future dispersant s. E ven if the dispersants themselves were completely non-toxic, the proposed dispersed oil toxicity test and associated pass/fail threshold could prevent all dispersants from being listed and available for use in a spill response . The Proposed Rule covers activities broader than mere scheduling criteria for dispersants. The added complexity resulting from the Proposed Rule in auth orization, monitoring and product listing, in addition to unfounded and overly corrective assumptions about dispersant effects described in the Proposed Rule, will not assist the Unified Command during an oil spill . Instead, it will likely complicate and limit its ability to respond in the most effective manner to ensure the health and safety of people and minimize environmental and community impacts. Also, prohibiting the application of dispersants during an oil spill response or delaying their use will significantly increase the economic and environmental damage of oil spills.

API and NOIA encourage the Agency to review the comments contained herein , incorporate the corresponding recommendations in a supplemental proposed rule, and provide the public and other Federal government and state stakeholder agencies with an opportunity to provide another round of comments be fore implementing a final rule. We also recommend that EPA initiat e a Negotiated Rulemaking under the Negotiated Rulemaking Act of 1990 to establish a negotiating committee under the Federal Advisory Committee Act to develop draft regulatory text on a consensus basis for adoption and implementation as a proposed supplemental proposed rule by EPA. A balanced negotiating committee made up of response experts – from the Federal Agencies who oversee oil spill response and from industry and relevant third-party experts – would add value to further Proposed Rule revisions and ensure that Subpart J is aligned with the latest scientific consensus on, and avoid an artificial bias against, the use of dispersants. As an example, Negotiated Rulemaking was successfully used to implement vessel and facility response planning requirements under the Oil Pollution Act of 1990. At a minimum, we believe that E PA should facilitate a discussion between the response experts before finalizing the Proposed Rule.

We first provide general comments followed by detailed comments on a section -by-section basis for EPA's consideration below.

GENERAL COMMENTS

Industry Shares EPA's Goal Of Establishing Sound Dispersants Policy Grounded In Science and Executable in Real-World Spill Scenarios

In every operation, the goal of industry is to never have an oil spill, and it invests extensive time and effort to prevent spills from occurring. Industry and government have a shared interest in a unified, coordinated, and proactive approach that allows us to protect our common values. These values include the health and safety of our citizens, sensitive ecosystems, the local business community and economy, and the vitality and sustainability of tourism and other key community industries, such as fishing.

To achieve a well-managed response and the protection of our common values, the primary guiding principle is to stop the source of a spill as quickly as possible with the goals of ensuring the health and safety of people and minimizing environmental and community impacts. In many spill circumstances, dispersants play a critical role in achieving these goals, perhaps most importantly by reducing risks associated with Volatile Organic Compounds (VOCs) in the immediate vicinity of the spill that could be both a human health and safety issue, and should continue to be a response option promptly available to the Unified Command. The Proposed Rule could jeopardize the protection of the common values by potentially precluding even completely non-toxic dispersants from being listed due to the proposed dispersed oil toxicity tests and associated pass/fail thresholds. Scientific studies have shown that approximately 97.5% of the TPH constituents in dispersed oil toxicity tests come from the oil (not the dispersant), so the proposed laboratory tests will measure the toxicity of oil (not the dispersant).[1] If the standardized oils (i.e., ANS and IFO-120) required to be used in the proposed test have toxicity values (LC50) in water accommodated fractions (WAF) less than 10 ppm, then no dispersant, no matter its individual toxicity, will meet the threshold values for listing.

Moreover, the creation of arbitrary thresholds for passing such tests, especially when EPA has shown that modern dispersants are of low toxicity, is not supportable and it is not clear whether any dispersant will be approved for the NCP Product Schedule when both toxicity and effectiveness tests are required. Specifically, setting an acute toxicity threshold of dispersants plus oil at an LC50 of 10 ppm has a high probability of eliminating an effective dispersant from the list, i.e., any dispersant that passes the baffled flask effectiveness test. We know from previous testing by industry that many (or even most) crude oils by themselves have LC50's below 10 ppm. Of more relevance is the use of toxicity results to assist in a NEBA-based discussion of the relative merit of using a particular dispersant in favor of another. Similarly, the attempt to link lab test results to decisions relating to pre-authorization and use is misplaced since laboratory tests are useful in understanding differences between products, not their real world performance.

The Proposed Rule indicates that the Agency " *does not state or imply* " that one response option is endorsed over another which is a practice shared by industry. However, the preponderance of additional requirements for dispersant u se will likely have the unintended consequence of prohibiting, or delaying the approval of, dispersant use in a timely manner regardless of whether they may be the best, or only, response option. Further, the language contained in the preamble suggests that the Agency considers dispersants as a last priority, rather than giving them equal consideration with all other appropriate response options. Additionally, API and NOIA are concerned that the Proposed Rule may cause chemical

[1] Scientific reasoning that the dispersant only contributes ~2.5% of the TPH toxicity is based on the following. In a 1:20 disp ersant-to-oil ratio (DOR), the dispersant proportion is 5%. The solvent component of modern day dispersants is roughly half, which mean s the overall TPH contribution from the dispersant is ~2.5%. As a result, the toxicity is dominated by the oil constituents.

manufacturers to either disco ntinue the production of effective, commercially -available products or discontinue research into the development of the next generation of dispersants.

A Net Environmental Benefit Analysis Process Ensures Response Choices that Minimize the Impacts of Oil Spills on People and the Environment

Any changes to the rule must take into account the Net Environmental Benefit Analysis (NEBA)[2] process. When a spill occurs, the OSC must make time-critical choices about which response options to implement immediately and effectively to manage the potential impacts of the spill. The NEBA process helps the OSC select choices that minimize the impacts of the spill on people and the environment. NEBA is a consensus-based planning tool that is used to bring natural resource trustees, regulators and stakeholders together to address resource-management decision-making needs for an oil spill response. It provides a structured, scientific method to evaluate the likely net reduction, or possible increase, in environmental and socio-economic impacts resulting from the implementation candidate response options as compared to the natural recovery or "do nothing" alternative. An assessment of the trade-offs associated with each option is then conducted taking into consideration the potential impacts to sensitive environmental resources and local community values. The NEBA process determines which response option or combination of options can best reduce the overall impact of spilled oil to common values and promote the fastest overall ecosystem recovery. Of particular note, the NEBA process considers the potential effects to a wide range of concerns, ranging from likely environmental effects (i.e., deep marine to terrestrial, etc.) to possible economic consequences. When initially considering the potential outcomes, no individual area receives a greater priority over another. Instead, all are weighed from an equal footing and only after evaluating all the likely effects is a determination made regarding the response option that offers that greatest environmental benefit. In so doing, decision-makers ensure that an unbiased process is used, with input from relevant stakeholders, to determine the best course of action. This process has been adopted all over the world to identify response methods that will yield the greatest benefit with the least impact.[3]

The NEBA process can be used in both oil spill preparedness planning and response. In the United States, a formal NEBA process is conducted during the planning phase at the Area Committee and Regional Response Team (RRT) levels with input from state and Federal participants to determine the benefits and limitations from using each response technology within their individual areas of responsibility. In most offshore regions of the U.S., this is accomplished in the form of the Consensus-Based Ecological Risk Assessments (CERA) process (NEBA equivalent) used by the U.S. Coast Guard (USCG)[4] and supported by the National Oceanic and Atmospheric Administration (NOAA), EPA, other Federal and state agencies, and academia. More than twenty workshops were held in various locations around the continental U.S., Caribbean, and Alaska from 1995 to 2011 to compare the benefits and risks of various response options when considering resource trade-off decisions. All of the workshops resulted in final publications (available from USCG) that were delivered to the Area Committees and RRTs to assist with response planning. One example of how this CERA/NEBA process was used to inform dispersant use decision-making is summarized in several papers authored by regulators in the state of California.[5] The applicability of the CERA/NEBA process as a tool for facilitating dispersant decision-making during spill response and planning was also evaluated by NOAA and published in a 2008 paper.[6]

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[2] NEBA is similar to the Consensus Ecological Risk Assessment (CERA) process developed by the U.S. Coast Guard in the mid -1990s.

[3] See, *Incident management system for the oil and gas industry: good practice guidelines for incident management and emergency response personnel*, IPIECA, August 2014, http://www.ipieca.org/publication/incident-management-system-oil-and-gas-industry-good-practice-guidelines-incident-manage.

[4] Aurand, D., Walko, L., & Pond, R. (2000). *Developing consensus ecological risk assessments: Environmental protection in oil spill response planning*. Washington, D.C.: U.S. Coast Guard.

[5] Addassi, Y. N., & Faurot-Daniels, E. (2005). California oil spill dispersant plan -Achievement through cooperation. *Proceedings of the 2005 International Oil Spill Conference, 2005* (1), 433 -437. doi:10.7901/2169 -3358-2005-1-433; Addassi, Y. N., Sowby, M.,

In 2014, the practice of incorporating a NEBA review during Source Control drills in the Gulf of Mexico was implemented by RRT VI during several large scale spill response exercises,[7] which has fostered good cooperation between the agencies to focus on the goal of minimizing shoreline community and ecosystem impacts by selecting response options that can maximize the effective treatment of oil offshore. NEBA discussions during both of these recent spill drills resulted in the concurrence of the RRT to authorize dispersants (both aerially and subsea) to treat the offshore oil slick.

Government decision-makers and/or the Responsible Party can conduct a rapid NEBA during actual spill responses to compare and rank the strengths and weaknesses of different response options for the incident-specific conditions. Lessons learned following a spill are incorporated into future policies and plans. Ecological recovery studies of the 1996 *Sea Empress* spill in the UK demonstrate the benefit of dispersant application along the coastline to prevent persistent oil from stranding in shoreline habits.[8] A more recent publication examined the recovery rates for a tropical ecosystem during a controlled study of dispersed versus untreated shoreline oil slicks and concluded that better recovery occurred in the areas where dispersant was used.[9]

Regrettably, the Proposed Rule lacks a NEBA perspective. It incorrectly focuses on intrinsic, hazard-based criteria for substance listing and ignores the more appropriate risk-based trade-off decision process responders must consider during an oil spill response. It does not recognize that dispersants are only applied when regulators and response decision-makers deem them necessary to prevent more significant and harmful consequences. The Proposed Rule favors protecting pelagic and benthic species while putting critical habitats, such as the ocean's surface and near-surface zones, as well as sensitive shorelines, at greater risk to environmental damage. Modeling results presented at the 2005 International Oil Spill Conference concluded that:

> *"…the results of the modeling support the contention that there are more opportunities to save wildlife, shorelines, and sensitive habitats along the shore with dispersant use than there are risks of impacting water column biota. If the areas that would be impacted by surface oil are those where wildlife are concentrated, as they typically are near shore, and the water column impacts resulting from dispersant use would be offshore where water column biota are lower in abundance, the trade-off is much more heavily weighted toward dispersant use before oil comes near the shoreline. In fact, for medium or smaller spills, or even larger spills where sensitive species and life stages are not present, the impact of dispersant use on water column biota could be negligible."*[10]

Parker-Hall, H., & Robberson, B. (2005). Establishment of dispersant use zones in the state of California: A consensus approach for marine waters 3 –200 nautical miles from shore. Proceedings of the 2005 International Oil Spill Conference, 2005(1), 187 -191. doi:10.7901/2169-3358-2005-1-187.

[6] Mearns, A., & Evans, M. (2008). Evaluating the consensus ecological risk assessment workshop as a tool for facilitating decision - making during spill response and planning. *Proceedings of the 2008 International Oil Spill Conference Proceedi ngs, 2008*(1), 739-741. doi:10.7901/2169-3358-2008-1-739

[7] Freeport McMoran Source Control drill (April 30-May 1, 2014) and BP Source Control drill (Nov 5-6, 2014)

[8] Lunel, T., Rusin, J., Bailey, N., Halliwell, C., & Davies, L. (1997). The net environmental benefit of a successful dispersant operation at the Sea Empress incident. *Proceedings of the 1997 International Oil Spill Conference, 1997* (1), 185 -194. doi:10.7901/2169-3358-1997-1-185

[9] Baca, B., Rosch, E., DeMicco, E. D., & Schuler, P. A. (2014). TROPIC S: 30 -year follow -up and analysis of mangroves, invertebrates, and hydrocarbons. *Proceedings of the 2014 International Oil Spill Conference, 2014* (1), 1734 -1748. doi:10.7901/2169-3358-2014.1.1734

[10] French McCay, et al., Modeling Fates and Impacts of Hypothe tical Oil Spills in Delaware, Florida, Texas, California, and Alaska Waters, Varying Response Options Including Use of Dispersants. International Oil Spill Conference Proceedings: May 2005, Vol. 2005, No. 1 pp. 735-740. Also available at http://ioscproceedings.org/doi/pdf/10.7901/2169-3358-2005-1-735.

For oil spills where dispersant use is contemplated, a typical NEBA will predict the potential impacts to aquatic organisms from an increased exposure to dispersed oil droplets. Dispersant use is typically restricted to water depths greater than 10 to 20 meters so that dilution will minimize exposure or impacts to the benthic community. Studies have shown the dispersed oil concentrations are rapidly diluted within the large volume of receiving water from a high of a few hundred parts per million to a few parts per million within a few hours. Consequently, exposure to the toxic effects of the dispersed oil will be of short duration and limited to those aquatic organisms that inhabit near surface waters. The NEBA will also estimate the reduction of impacts to waterfowl, shoreline ecosystems and coastal communities resulting from the reduction in, or elimination of, the quantity of floating oil. The trade-offs of a short term increase in exposure of selected aquatic organisms to dispersed oil versus an increase in oil impacting coastal environments and communities must be evaluated and take into account local community values to determine if dispersant use is warranted.

Given the challenges of implementing other response options, a decision not to have dispersants available to responders could result in oil stranding on shorelines where it may persist for an extended time. Effectively dispersed oil will biodegrade rapidly so that the majority of the oil will persist for a much shorter period of time. Reducing the persistence of the oil in the environment is critical to reducing the overall impacts.[11] This is not to say that water column organisms are less intrinsically important than surface or shoreline species but using NEBA can and does provide a critical framework for deciding what is best for the overall ecosystem. Important considerations to understand in assessing this tradeoff include:[12]

- The impacts to the water column are mitigated by dilution – that is the potentially toxic levels of dispersed oil are limited in duration and physical coverage/extent and effected populations (e.g., plankton) tend to be ones that recover quickly.
- Oil effectively dispersed in the water column will generally biodegrade after a few days or weeks while oil stranded in near shore and shoreline areas can persist for years.
- Near shore and shoreline areas are the nursery grounds for many species, even those that live offshore as adults, so protecting the near shore areas also protects organism that live as adults offshore.
- The bio density of near shore areas (swamps and marshes) is far greater than the bio density in pelagic waters offshore.
- Dispersants also protect surface dwelling organisms offshore, i.e., marine mammals, birds, and turtles, through removal of floating oil.
- Slicks on the water surface are far more likely to impact fish eggs and larva that tend to float at or near the water surface than quickly removing oil from the surface with dispersants.
- In certain geographies, slicks can impact mangroves and other shoreline vegetation that can subsequently die off and result in shoreline erosion that increases sediments in the near shore water column and potentially affects corals and other species.

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[11] Hazen, et al., Deep-Sea Oil Plume Enriches Indigenous Oil-Degrading Bacteria, Science, 330 (2010), pp. 204-208.

[12] (a) French McCay, D., and Graham, E. Quantifying Tradeoffs – Net Environmental Benefits of Dispersant Use. International Oil Spill Conference Proceedings: May 2014, Vol. 2014, No. 1, pp. 762-775.
(b) McManus, M.A., Woodson, C.B. Plankton Distribution and Ocean Dispersal. The Journal of Experimental Biology, 215 (2012), pp. 1008-1016.
(c) DeMicco et al., Net Environmental Benefit Analysis (NEBA) of Dispersed Oil on Nearshore Tropical Ecosystems: TROPICS – The 25th Year Research Visit, International Oil Spill Conference Proceedings, May 2011, Vol. 2011-282.
(d) Varela et al., The Effect of the "Prestige" Oil Spill on the Plankton of the N -NW Spanish Coast. Marine Pollution Bulletin 53 (2006), pp. 272-296.
(e) Addassi, et al., Establishment of Dispersant Use Zones in the State of California: A Consensus Approach for Marine Waters 3-200 Nautical Miles from Shore. International Oil Spill Conference Proceedings: May 2005, Vol. 2015, No. 1, pp. 187-191.

The Agency should consider the NEBA process and how and why dispersants are used before making changes to the Subpart J Regulations.

The Proposed Rule has the Practical Effect of Treating Dispersants as Toxic and either Prohibiting or Discouraging the Use of this Critical Response Tool

Dispersants are an essential tool in the responder's toolkit. They are products used in oil spill response to dilute the oil in the water column to enhance natural microbial degradation, a process where microorganisms remove oil from the environment.[13] All marine environments contain naturally occurring microbes that feed on and break down crude oil. Dispersants aid the microbial degradation by forming tiny oil droplets, typically less than the size of a period on this page (<100 microns), thus increasing the oil's surface area and making it more available for the petroleum-degrading microorganisms to consume.[14]

All oil spill response options have their place in the response toolkit because of the extreme variability of spill conditions. Dispersants become a critical response tool for larger spills far from shore; spills more distant from stockpiles of recovery and containment equipment; when weather and ocean conditions preclude the use of other options; or when weather conditions are predicted to become more severe because they can be rapidly applied by aircraft or subsea (at a high encounter rate) when wind and wave conditions prevent other vessel-based recovery options.

The Proposed Rule focuses on the potential risks of dispersants without recognizing the benefits and core purpose of dispersants, which is to rapidly dilute the spilled oil and enhance removal of the oil from the environment through microbial degradation, resulting in reduced VOCs and shoreline impacts. Ultimately, the purpose of dispersants is to reduce the overall toxicity and persistence of oil in the environment.

We believe that research showing that dispersants do not enhance the biodegradation of oil from the environmental is based on studies that used unrealistically high concentrations of dispersed oil. Prince and Butler only recently developed a biodegradation test method that compares dispersed oil biodegradation to biodegradation of a surface slick at representatively low concentrations (i.e., 2.5 ppm).[15] This research unambiguously shows the enhanced biodegradation of dispersed oil over allowing an oil slick to remain on the surface. Most prior dispersed oil biodegradation studies used oil concentrations that were two or more orders of magnitude too high.[16]

Dispersants have proven to be an effective way to respond to an oil spill, most notably in those situations that prohibit, or otherwise limit, the effectiveness of other response measures. We submit that all response options should be contemplated based on their individual, or combined, ability to respond to

[13] American Society for Microbiology (ASM). (2011). *FAQ: Microbes & oil spills.* Retrieved from http://academy.asm.org/images/stories/documents/Microbes_and_Oil_Spills.pdf; Personna, Y., Boufadel, M., & Zhang, S. (2014). Biodegradation of dispersed Endicott oil in controlled experiments. Proceedings of the 2014 International Oil Spill Conference, 2014(1), 1126-1140. doi:10.7901/2169-3358-2014.1.1126; McFarlin, K., Leigh, M., & Perkins, R. (2014). Biodegradation of oil and dispersed oil by arctic marine microorganisms. Proceedings of the 2014 International Oil Spill Conference, 2014(1), 300-317. doi:10.7901/2169-3358-2014-1-300317.1.

[14] Hazen, T. C., Dubinsky, E. A., DeSantis, T. Z., Andersen, G. L., Piceno, Y. M., Singh, N., Jansson, J. K., Probst, A., Borglin, S. E., Fortney, J. L., Stringfellow, W. T., Bill, M., Conrad, M. E., Tom, L. M., Chavarria, K. L., Alusi, T. R., Lamendella, R., Joyner, D. C., Spier, C., Baelum, J., Auer, M., Zemla, M. L., Chakraborty, R., Sonnenthal, E. L., D'haeseleer, P., Holman, H. N., Osman, S., Lu, Z., Van Nostrand, J. D., Deng, Y., Zhou, J., & Mason, O. U. (2010). Deep-sea oil plume enriches indigenous oil-degrading bacteria. *Science*, *330*, 204-208. doi:10.1126/science.1195979

[15] Prince, R.C. and Butler, J.D. A protocol for assessing the effectiveness of oil spill dispersants in stimulating the biodegradation of oil. Environ Sci Pollut Res Int. 2014 Aug;21(16):9506-10. doi: 10.1007/s11356-013-2053-7. Epub 2013 Aug 13.

[16] Lee, K., Nedwed T., Prince R.C., Palandro D., Lab tests on the biodegradation of chemically dispersed oil should consider the rapid dilution that occurs at sea. Mar Pollut Bull. 2013 Aug 15;73(1):314-8. doi: 10.1016/j.marpolbul.2013.06.005. Epub 2013 Jun 27.

discharged oil, either on or below the water's surface, in a manner that causes the lowest negative effect on the environment and promotes the most rapid recovery of the affected ecosystem.

Laboratory Dispersant Efficacy Tests do not Predict How Well Dispersants Will Work in the Wide Range of Real World Spill Conditions

Although laboratory dispersant effectiveness testing can be used as a comparative screening tool for different dispersants under laboratory conditions, the Proposed Rule appears to suggest these results will be representative of actual effectiveness when applied to oil spills on open water. However, these results do not translate to the wide range of field conditions, oil types and weathering degrees in real world oil spills. Specifically, the static conditions in a lab, where researchers conduct experiments with known quantities of oil and dispersant, cannot be replicated at sea. Although the amount of dispersant that is to be applied is calculated based on a desired initial dispersant-to-oil ratio, the quantity and characteristics of oil during a real spill is an estimate, making any definitive, quantitative effectiveness estimation problematic. Localized determinations of the relative effect of surface dispersant application may be evaluated qualitatively using visual cues or semi-quantitatively using ultraviolet fluorometry, which is sufficient for operational effectiveness assessments. Additionally, much of the detailed spill monitoring specified by the Agency in the Proposed Rule has no bearing on dispersant effectiveness monitoring and cannot be conducted in a time-frame that will support day-to-day, near real-time operational decision-making.

We suggest the language in the Proposed Rule be modified to clarify the laboratory efficacy testing is for comparative purposes only and that the pass/fail criteria be established at a level that will only eliminate the least effective dispersants from being listed. Some dispersants that only exhibit average effectiveness in the laboratory tests may be very effective in certain situations in the field and the Response Community needs to maintain as many options as possible to select the most appropriate response option for a given set of conditions and circumstances.

Dispersant Toxicity Tests do not Accurately Simulate Real World Exposures.

API and NOIA acknowledge that there may be benefits to a so-called "gate-keeper" toxicity test with dispersant alone to ensure that any dispersants that might contain highly toxic components would not be accepted for listing on the Product Schedule. However, it is unclear what rationale was used for the dispersed oil toxicity tests stated in the Proposed Rule.

The relevant factors in dispersed oil toxicity include the concentration of dispersed oil, or oil compounds, in the water and the duration of exposure. The fundamental basis of aquatic toxicity testing is to create a set of concentrations and expose the animals for a duration where observable effects can be measured. As a result, laboratory exposures are useful for data collection and generating toxicity curves; however these conditions are not representative of the real world where the natural processes of rapid dilution and biodegradation act on the dispersed oil to dramatically reduce the exposure conditions [17]. Using acute 48- or 96-hour LC_{50} toxicity tests with dispersed oil mixtures (to calculate LC_{50} values) is considerably different from the likely exposure regime experienced by marine organisms in the open sea following an oil spill. For this reason, the Chemical Response to Oil Spills: Ecological Effects Research Forum (CROSERF)[18] was assembled to develop new dispersed oil toxicity testing protocols that account for rapid dilution. Academia, regulators and industry funded a ten year program through CROSERF to

[17] Lee, K., Nedwed, T., & Prince, R. (2011). Lab tests on the biodegradation rates of chemically dispersed oil must consider natural dilution. *Proceedings of the 2011 International Oil Spill Conference, 2011* (1), 245. doi:10.7901/2169-3358-2011-1-245
[18] Aurand, D., & Coelho, G. (2005). *Cooperative aquatic toxicity testing of dispersed oil and the chemical response to oil spills: Ecological effects research forum (CROSERF)*. Ecosystem Management & Associates, Inc., Technical Report 07-03. Lusby, MD.

develop new testing methods that are more aligned with real -world field c onditions. This data is now available via the NOAA Dtox database.[19]

Results from dispersed oil monitoring during the Deepwater Horizon (DWH) spill is clear evidence of the rapid dilution occurring after surface dispersant application [20] and subsea dispersant injection .[21] Further evidence of the rapid dilution of dispersed oil in the field is found in the analytical results from several field trials conducted in the North Sea to characterize the fate and transport of dispersants applied to an open ocean oil slick. These field experiments and the subsequent chemical characterization condu cted in 1994,[22] 1995,[23] and 1996[24] showed a rapid (within one hour) dilution of dispersed oil concentrations following dispersant application. This same effect was observed during the DWH response following surface dispersant application.[25] Since the proposed EPA dispersed oil testing does not take into account dilution, it is more likely that reported toxicity values at a rtificially high laboratory concentrations, not representative of real -world dilution exposures, will lead to misinterpretation of these tes t results and possibly preclude the use of dispersants altogether.

We strongly suggest that the laboratory toxicity test be limited to dispersants only to screen out the more toxic products and that toxicity testing for dispersed oil be explicitly elimin ated from the Rule. It is very likely that all dispersants will be unable to pass the dispersed oil toxicity test as existing oil toxicity data indicates the test oils alone will be unable to pass the test.

In the context of NEBA (and its forebear, th e Consensus Ecological Risk Assessment, or CERA), scientists understand and accept that marine organisms may experience localized, transient toxic effects from exposure to dispersed oil, but these local effects are offset by the benefit of dispersant use i n preventing or minimizing oil drifting ashore and causing severe and long -term harm to more sensitive coastal resources and communities . While knowledge of the potential for toxic effects to marine organisms is useful for conducting a NEBA, it cannot be t he sole indicator that dictates the use of dispersants. All resources -at-risk as well as coastal communities must be considered in the response option decision-making process, with the emphasis on coordinating a response that will result in the least overall impacts while promoting the most rapid overall recovery of the affected ecosystem.

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[19] Bejarano, A., Chu, V., Dahlin, J., & Farr, J. (2 014). Development and application of dtox: A quantitative database of the toxicity of dispersants and chemically dispersed oil. *Proceedings of the 2014 International Oil Spill Conference* , *2014*(1), 733 -746. doi:10.7901/2169-3358-2014.1.733

[20] Operational Science Advisory Team (OSAT 1). (2010). *Summary report for sub-sea and sub-surface oil and dispersant detection: Sampling and monitoring. Unified Area Command, New Orleans*. Retrieved from: http://www.dep.state.fl.us/deepwaterhorizon/files2/osat_report_17dec.pdf .

[21] Coelho, G., Aurand, D., Essex, L., Parkin, A., & Robinson, L. (2012). *Monitoring subsurface dispersant injection during the MC252 incident, volume 1*. Lusby, MD, USA: Ecosystem Management & Associates, Inc.

[22] AEA Technology. (1994). *International calibration of laboratory dispersant test methods against sea trials* . Oxfordshire, UK: AEA Technology.

[23] AEA Technology. (1995). *International calibration of laboratory di spersant test methods against sea trials. Field trial report. July 1995 sea trials*. Oxfordshire, UK: AEA Technology ; Jones, M., & Petch, S. (1995). A report on the analysis of hydrocarbons in sea waters and associated samples from trial oil spills off east ern England, July 1995. Newcastle upon Tyne, UK: University of Newcastle upon Tyne.

[24] Strøm-Kristiansen, T., Hokstad, J. N., Lewis, A., & Brandvik, P. J. (1997). *NOFO 1996 oil on water exercise – analysis of sample material. SINTEF Data report number STF66 A97050*. Trondheim, Norway: SINTEF ; Coelho, G. M., Aurand, D. V., Petch, G.S., & Jones, D.M. (1998). Toxicity bioassays on dispersed oil in the North Sea: June 1996 field trials. Ecosystem Management & Associates, Inc., Report 96-02. Purcellville, VA: Ecosystem Management & Associates, Inc.

[25] Huber, C. A., Steen, A., & Parscal, B. (2014). Does wave height matter for effective surface dispersant application? *Proceedings of the 2014 International Oil Spill Conference, 2014* (1), 747-761. doi: 10.7901/2169-3358-2014.1.747

During an Oil Spill, Operational Monitoring Should Focus on How the Selected Response Options Mitigate the Effects of the Oil on the Environment

Following an oil spill, the primary emphasis of operational response monitoring should be directed toward the response effort and the effect that those efforts have on reducing the net negative environmental consequences. The Agency proposes the requirement to "*monitor agent use in the environment*." API and NOIA submit that operational monitoring should not be focused on monitoring the agent (i.e., dispersant) use, but rather on monitoring the overall changes to the characteristics, transport, and fate of the oil within the ecosystem. We also submit the Agency should acknowledge that the primary issue in a spill scenario is oil in the environment (not dispersants in the environment) and that operational monitoring efforts should concentrate on evaluating how well the response option (or options) mitigates the negative effects of the oil on sensitive environmental resources.

EPA proposes monitoring that will provide a comprehensive assessment of the environment. However, the monitoring actions required in the Proposed Rule go beyond those required to inform operational decision-making and cross into the realm of Natural Resource Damage Assessment (NRDA) monitoring. One reason for this is that NRDA monitoring requires days to years before opinions on damages are made. Operational decisions require real or near-real time data. Damage assessment monitoring should not fall under the purview of spill response decision-making, and any monitoring of response options should be done solely to ensure the effectiveness of that response technique.

The Proposed Rule includes additional surface dispersant effectiveness monitoring requirements beyond those established in current practices, with no justification of how this additional monitoring burden will enhance operational response decision-making. Further, the additional requirements do not differentiate between surface and subsurface use of dispersants. As a result, the Proposed Rule would apply an unnecessary set of requirements in the event of surface dispersant usage in lieu of those monitoring techniques outlined in the USCG *et al., Special Monitoring of Applied Response Technologies* (SMART) protocols.[26] Additionally, the monitoring regime outlined in the Proposed Rule deviates substantially from that recommended by the National Response Team (NRT) in its *Environmental Monitoring for Atypical Dispersant Operations: Including Guidance for Subsea Application and Prolonged Surface Application,*[27] which also advocates the adaptation of the SMART monitoring regimen. These additional monitoring requirements, if adopted, may divert valuable resources away from the response effort, potentially prolonging the spill event. They may also preclude the use of dispersants in many cases if required to be in place as a condition of authorization, as it can take many days to assemble the equipment, vessels, and scientific experts.

During a subsea dispersant application, the Agency proposes an exhaustive set of environmental monitoring parameters that include *in situ* oil droplet size distribution analysis, *in situ* fluorometry, fluorescence signatures, dissolved oxygen, total petroleum hydrocarbons, carbon dioxide, methane, heavy metals, turbidity, water temperature, pH, and conductivity. As stated previously, we concurs with monitoring that supports effective, operational decision-making, but argues that most of these parameters will not provide any information on the effectiveness of the subsea dispersant application, will unnecessarily require additional scientific resources and will delay the reporting of key operational monitoring parameters. Spill response effectiveness monitoring should be based on simple, rapid assessments that can provide near real-time information on the progress of the response. For this reason, we believe that any mandated operational monitoring should be both incident-specific and designed to integrate into the other operational activities taking place. As such, the Responsible Party

[26] These protocols were jointly developed by the U.S. Coast Guard, National Oceanic and Atmospheric Administration, U.S. Environmental Protection Agency, Centers for Disease Control and Prevention, and Minerals Management Service.
[27] See http://www.nrt.org/production/nrt/nrtweb.nsf/AllAttachmentsByTitle/SA-1086NRT_Atypical_Dispersant_Guidance_Final_5-30-2013.pdf/$File/NRT_Atypical_Dispersant_Guidance_Final_5-30-2013.pdf?OpenElement

shares in the responsibility with the OSC in determining what monitoring needs to be done in ac cordance with the region -specific pre -approval and planning process. Comprehensive, detailed environmental monitoring is not an appropriate topic for this Proposed Rule.

The Agency's Economic Review Fails to Recognize Significant Costs of the Proposed Rule.

The Regulatory Impact Analysis (RIA) conducted by the Agency does not estimate all of the potential policy scenarios, and the economic impacts of those policies, that are issued in the Proposed Rule. Therefore, the RIA neglects to provide the public with a full description of the proposal's impact. Additionally, EPA did not provide any quantified social benefits that could result from implementation of any part of the Proposed Rule. An independent assessment [28] of the proposal (see Attachment A) has found that including the impacts of the requirements neglected by the Agency in the RIA would increase the cost of the proposal three times. In addition to underestimating compliance costs and administration costs for both industry and the federal governmen t, the RIA supporting the proposal fails to calculate the costs associated with the immediate market imbalances that could occur if products are removed from the market, either temporarily or permanently.

Beyond the market imbalance, removal of dispersants from the toolbox could result in additional clean-up costs that are not considered in the RIA. As noted in Attachment A, "To the degree this rulemaking reduces chemical and biological agent availability and innovation, the nation will have less tools av ailable to fight oil spill damage." A possible outcome of this limitation "is that future oil spill damages would be more severe than they would be if the rule did not occur." EPA did not quantify the costs associated with the possibility that the rule w ould limit the ability of the Response Community to keep oil from reaching shorelines or damaging ecosystems to a greater extent than if dispersants were readily available. We believe additional costs could extend to costs associated with environmental cla ims, increased response costs due to the need for shoreline cleanup, and other costs of environmental damages to sensitive shorelines, birds, and lost economic and recreational opportunities. In fact, EPA confuses the social benefits of avoiding an oil spill or having an appropriate response to a spill with the social costs of a spill, such as lost tourism revenue and other environmental damages. Furthermore, the Agency underestimated the costs to small businesses, which resulted in the false conclusion t hat small businesses would not be materially affected by the proposed ruling. Using realistic estimates, the attached third-party report finds that 43 percent of the firms identified by the Agency would face compliance costs in excess of three percent of a nnual sales and risk of closure. The Agency should amend the methodology and deficiencies found in the RIA, as discussed here and in the attached report.

We explicitly incorporate the comments and recommendations addressing the RIA provided by the independent assessment (see Attachment A).

The Proposed Rule Conflicts with Current Facility and Vessel Oil Spill Response Plans

Effective September 30, 2009, facility and tank vessel plan holders were required to update their response plans to include planning for responding to discharges using dispersants under a USCG rulemaking. Specifically, dispersant equipment is required in areas where it has been pre-determined that dispersants would be a viable oil spill mitigation technique and pre-authorizations have been established. Previously, with regard to dispersants for facilities located in areas with pre-authorization, a mechanical recovery equipment credit for up to 25% could be obtained for certain persistent oils.

[28] The independent assessment was prepared by the Policy Navigation Group (PNG), which consists of a diverse team of senior policy analysts with experience at Office of Management and Budget in the White House, managers, economists, engineers, lawyers and skilled researchers, with particular expertise in both private and public sector environmental management strategies including benefit-cost analysis and life-cycle assessments. See http://www.policynavigation.com/index.php?page=bios for more information about PNG.

However, since 2009 facilities handling certain persistent oils in areas with pre-authorization must ensure, through contract or other approved means, the availability of dispersant response resources and the commencement of dispersant-application operations within 7 hours of the decision by the OSC to use dispersants. Dispersant response resources must include: (1) sufficient volumes of dispersants to meet specified regulatory requirements, (2) dispersant application platforms and systems sufficient to meet planning standards, and (3) trained personnel.

Similarly, vessels transporting certain persistent oils in areas where dispersants are pre-authorized have been since 2009 required, through contract or other approved means, to have dispersant-related response resources meeting dispersant planning standards, including appropriately trained dispersant-application personnel. The dispersant resources must include the identification of each primary dispersant staging site used by each dispersant-application platform and the corresponding distance to the associated stockpile. Similar requirements were adopted for non-tank vessels in 2013.

There are also aerial tracking requirements under the current response plan requirements, and there are now pre-authorization zones for the use of dispersants outlined in the applicable Area Contingency Plan throughout the nation for every state except for Alaska and Connecticut whereby the OSC has the authority to expedite the authorization for the use of dispersants. Although there are no specific dispersants requirements under Bureau of Safety and Environmental Enforcement's (BSEE) response plan requirements for offshore facilities, it is clear based on experience with the *Deepwater Horizon* that dispersants are a key response tool that must be available on an expedited basis to respond both to a spill on the surface of the water and subsea in case of a blowout.

The Proposed Rule, however, appears to directly conflict with Coast Guard response plan requirements and raises a host of questions related to the continued validity of facility and vessel response plans. It is unlikely that any dispersants will pass both the efficacy and toxicity tests required by the Proposed Rule as a precursor to being listed on the NCP Product Schedule. For example, the EPA proposal may have the practical effect of virtually eliminating the entire stockpile of available dispersants nationwide that will be able to be listed on the NCP Product Schedule following full implementation of the Proposed Rule. In addition, it could have the practical effect of eliminating the usefulness of preauthorization zones. This could result in invalidating response plans.

Furthermore, the Proposed Rule, if implemented , would require significant additional requirements for information needed an d monitoring required before dispersants use approval is granted , which may prevent a plan holder from being able to deploy dispersants in the desired/required amount of time. For some spills it may mean that dispersants cannot be used at all due to limit ed window of opportunity. Thus, the Proposed Rule raises significant issues related to USCG and BSEE facility and vessel oil spill response plan compliance and could result in unnecessary and increased environmental damage and corresponding spill claims a s a result of increased damages. In this regard, we note that although the President delegated to EPA the responsibility for the amendment of the NCP, the Proposed Rule may extend beyond the limits of EPA's authority under that delegation or the applicabl e law. Furthermore, amendments to the NCP must be coordinated with members of the NRT , including the USCG and other relevant agencies, prior to publication for notice and comment. 40 C.F.R. § 300.2. It appears this coordination may not have been accompl ished sufficiently to avoid the conflicts discussed above. Accordingly, we recommend that EPA further coordinate with the USCG and BSEE to ensure that the Proposed Rule does not adversely affect USCG and BSEE response planning procedures and requirements.

<u>Agency Review of Public Comments</u>

Due to the significance of this rulemaking to the industry and to other Federal government agency stakeholders involved with spill response, it is critical the Agency provide adequate time for these other

agencies to review and provide feedback on comments submitted by the public for EPA's benefit and consideration. Accordingly, we strongly urge the Agency to seek and consider comments from the other relevant Federal government agencies on the public's comments befor e it decides on how to handle all input received in response to the Proposed Rule.

Availability of EPA Testing Data

It has come to the attention of API and NOIA that the Agency may have testing data related to dispersants that may not have been released to the public. If that is the case then the Agency should make that data publicly available as soon as possible in some form of a supplemental notice to this Proposed Rule. Supplemental comments should then be requested to ensure that the public has an opportunity to provide additional informed comments for EPA's consideration before finalizing the Proposed Rule.

DETAILED COMMENTS ON THE PROPOSED AMENDMENTS TO 40 CFR PARTS 110 AND 300

Our specific section -by-section comments on the Proposed Rule are presented as follows . These comments conform to the organization of the Proposed Rule on a section-by-section basis.

I. § Section 300.5 Definitions

A. *The definition of burning agents should not include ignition devices*.

Burning Agents. The proposed defi nition reads: " *Burning agents are additives that improve the combustibility of the materials to which they are applied through physical or chemical means* ." In the preamble (80 Fed. Reg. 3385) an example is given of a helitorch system, stating that the mate rials in the helitorch would be considered burning agents. Additionally, the Agency is seeking comment on whether it should add ignition devices to the definition of burning agent.

The proposed defin ition seems to combine burning agents (materials that actually change the combustibility of the material they are added to) and ignition agents (materials used to start combustion, as is the case with the helitorch). This confuses the question around including ignition devices within the definition of burning agents which, if happens, could lead to administrative pauses in requesting a test burn, or any burn during an emergency phase of a response.

Igniters are used to initially heat a small portion of the oil to release vapors that can be ignited. The initi al process of ignition is not the same as improving overall combustibility of an oil. For example, when starting a campfire, a match does not change the combustibility of the wood; it simply provides the energy to start the combustion process. Similarly, a n in-situ burn ignition system provides energy to heat and vaporize some of the oil so that it can start to combust. For this reason, we submit that ignition devices (both ignition liquids and equipment) should not be included in the definition of a burning agent.

We recommend the Agency adopt the following definitions:

> *Burning agents are additives that change the combustibility of the materials to which they are applied through physical or chemical means prior to ignition.*
>
> *Ignition agents are additives that ignite, or initiate combustion of the materials to which they are applied.*

B. *EPA's definition of dispersants does not fully define dispersants.*

Dispersants. The EPA proposed definition of dispersants in 300.5 should be expanded to include an explanation of why the production of smaller droplets is important. The following language should be added to the end of the proposed definition: *"The formation of small droplets is intended to allow the oil to be rapidly diluted in open ocean environments t o non -harmful concentrations and to promote rapid biodegradation of the oil by naturally occurring microbes."*

C. *Oil-mineral aggregate ("OMAs") are not sorbents*

Sorbents. API and NOIA offer several comments on the definition of sorbents in 300.5, as follows:

- We support the addition of clay as an example of an inorganic/mineral compound, especial ly since commercial products made from clay are currently on the market.
- The Agency expresses concern that sorbents made from particulate matter may be used as sinking agents. Many inorganic/mineral compounds that are quite dense are commonly used on terrestrial spills on roads, pavements and other hard surfaces. The rule should not limit their use for these situations.
- Very fine clays (like those used in OMAs), wh ile still nominally denser than water, will still suspend in the water column because the particle sizes are very small. The rule needs to acknowledge that OMA's are not sinking agents. If the Agency is concerned that inorganic/mineral sorbents are being u sed as sinking agents, then their use should be explicitly prohibited for use as a sinking agent. Uses for other purposes should be allowed.
- EPA should not limit how sorbents in particulate form are used. The use of OMA technologies is not to sorb oil but to promote dispersion and biodegradation. The purpose of the technique is not the same as for sorbents and should not be regulated as such. OMA technologies should be excluded in the definition of sorbents and should have separate testing/authorization requirements.
- While most sorbents are collected and recovered from the environment, some natural organic sorbents are spread over oil on habitats where the oil cannot be easily removed. The sorbent ties up the oil to reduce further migration and to limit the exposure to flora and fauna. The Agency should not discourage this application as it is sometimes considered a "best practice" for certain oiled habitats. As such, the Agency should either remove the phrase, "generally collected and recovered…" or add lang uage to indicate that loose, organic sorbents used in wetlands may not be recovered.

D. *Saltwater should be defined in the Proposed Rule*

Saltwater. In column 2, 80 Fed. Reg. 3389, of the preamble and subsequent sections the term "saltwater" is used. It is recommended that saltwater be defined in the definitions in 300.5 since the Proposed Rule states that dispersants are only allowed in saltwater environments. There is no definition of saltwater offered in the proposal. Dispersants have been shown to work a t salinity much lower than the typical 35 ppt ocean concentrations and effectiveness at salinity as low as 14 ppt has been reported. A definition where dispersants can be used needs to be provided, taking into consideration the data on dispersant performance in brackish or estuarine environments.

II. <u>Section § 300.910 Authorization for Agent Use</u>

A. *The Unified Command (UC) determines the best use of response assets on a daily basis.*

EPA should not establish a policy or rule that mechanical recovery is always the priority choice of countermeasures to use in a response. In the preamble for this section (column 1, 80 Fed. Reg. 3387), the Agency makes the following statement:

> *"During the DWH response, a priority countermeasures scheme was established to first use mechanical recovery via skimming/booming or in-situ burning followed by subsea dispersant and lastly surface dispersant use."*

This may have been the conceptual prioritization from the perspective of EPA, but in reality dispersants played a much larger role and it was the responsibility of the UC to determine the best use of response assets on a daily basis to account for weather conditions, VOC suppression, oil slick locations, response equipment availability, etc. Every spill is different and specific references to a single spill event can indirectly and inappropriately be perceived as a recommended practice and should be avoided. In any spill scenario, all response tools must be used to their maximum effect. The Agency's unprecedented use of "Directives" during the DWH incident severely limited the use of dispersants and circumvented the NEBA decision-making process allowing significantly more oil reaching the water's surface and, subsequently, the shoreline. This resulted in additional environmental impacts with no documented benefits to the marine ecosystem.

The Agency should reconsider this statement and prioritization as it puts unneeded complexity into the response decision-making process and inappropriately prioritizes mechanical recovery over other response options. The effectiveness of mechanical recovery can be limited in large, offshore spills as a result of typical environmental conditions, such as wave action/height or wind conditions as well as the extensive surface area that must be covered. Further, the Agency's artificial prioritization of this technique over other viable response options is contrary to the NEBA concept to determine the best blend of response options (with no pre-determined or pre-existing bias) to achieve the best overall protection and recovery for the ecosystem.

B. *Decision-making plans are also important to response decision-making.*

Later in the same paragraph referenced above, the Agency makes the following statement:

> *"The Agency believes that preauthorization or expedited decision making plans are critical elements of contingency planning activities."*

We strongly agree that preauthorization and expedited decision-making are important.

C. *Proposed changes involving preauthorization planning will increase the workload of all Regional Response Teams (RRTs) and Area Committees (AC) and restrict their flexibility.*

EPA should not restrict RRT or OSC activities related to preauthorization plans in areas where preauthorization is desirable and applicable. At the end of the same paragraph (column 2, 80 Fed. Reg. 3387), the Agency makes the following statement:

> *"The Agency believes these proposed revisions to the authorization of use provisions will assist OSCs, RRTs, and ACs in their advanced planning activities as they consider response methods that result in the greatest environmental protection."*

We agree with the intent to assist the OSCs and RRTs. However, the added complexity resulting from the Proposed Rule in authorization, monitoring and product listing, in addition to the overly conservative assumptions about dispersant effects described in the Proposed Rule, will not assist these entities.

Instead, it complicates and limits their ability to respond. The Agency must provide specific examples to show how this rule ass ists responders, particularly in relation to the OSC duties of making timely operational decisions, especially those decisions based on using the concept of NEBA to guide a response.

The existing rule states that the RRTs and Area Committees should evaluate the desirability of using various agents as part of their planning processes and state that "*RCPs and ACPs shall, as appropriate, include applicable preauthorization plans....* " Under the current rule, most RRTs did, in fact , create preauthorization plans, and those plans did include appropriate oceanographic and geographic limits . In practice, only Region VI used those plans to mitigate oil spills, and the Region VI OSCs, RRTs, and Area Committees became very familiar with their use . Policies and practic es for the use of the RRT VI preauthorization plan were created by the appropriate member agencies without regulatory requirements and evolved to account for changes in information technology.

The proposed language states, "*RRTs and Area Committees shall address in a preauthorization plan…*" This becomes a mandate for all RRTs and Area Committees, including those areas that may have little interest or need for such plans. In fact, some RRTs and Area Committees do not even have water bodies where dispersants could be used. Most of the factors that are required to be considered by the Proposed Rule are already addressed in preauthorization plans (e.g., water depth, distance from shorelines). Some factors, however, such as the requirement that " *preauthorization plans must specify limits for the quantities and the duration of use*," for agents are new and have the effect of limiting the application and effectiveness of the pre authorization plans. This requirement wo uld suggest that RRTs must know in advance the si ze of the potential releases, the resources at risk and what strategies are available to protect them, the dispersability of the oil, and the optimum dispersant -to-oil application rate. The OSC's flexibility and the effecti veness of the response method are limited and not assisted by such requirements, and EPA has not justified that the proposed requirement is necessary in relation to existing requirements.

The existing rule also states that " *preauthorization plans may address factors such as* ..." However, the Proposed Rule states " *Preauthorization plans should document…* " and adds requirements for consideration of factors such as "inventory" and "manufacturing capability of available agents." Again, this requires additional effort on the part of RRTs and Area Committees, where such work is already typically being done by industry as part of their planning processes.

The net effect of most of the changes involving preauthorization planning is an increase in the workload of all RRTs and Area Committees while r estricting their flexibility in areas where preauthorization is desirable and applicable.

III. Section § 300.910(a) Use of Agents Identified on the Schedule on Oil Discharges Addressed by a Preauthorization Plan

A. *Adding more entities complicates the planning process.*

Section § 300.910(a) states that the RRTs and Area Committees shall address preauthorization plans. API and NOIA do not see any valid reason to add more entities to the planning process. In the preamble to this section, the Agency makes the following statement (column 1, 80 Fed. Reg. 3388):

> *"Therefore, the Agency is proposing to include RRTs as another planning entity with responsibility for developing preauthorization plans, and is requesting comment on this change, and on the advantages or dis advantages of keeping the development of these plans at the AC level."*

We believes that while the RRTs may provide more practical experience to the planning effort, the potential benefit would be overshadowed by the added complexity and bureaucracy. Unle ss the Agency can provide a sound rationale, we recommend against including the RRTs in the development of preauthorization plans.

B. *Area Committees should identify shoreline habitats that can benefit from the appropriate use of dispersants.*

In this same section of the preamble (column 3, 80 Fed. Reg. 3388), the Agency makes the following statement:

> *"The RRT's and ACs should identify the affected biological resources and their habitats likely to be negatively impacted, as well as those that are expected to benefit."*

API and NOIA support the requirement that A rea Committees should identify shoreline habitats that can benefit from the appropriate use of dispersants or other response methods. This will advance the use of the NEBA concept. Note that many of these resources are already identified in Area Contingency Plans.

C. *Preauthorization plans should provide the OSC sufficient authority to use chemical or biological response agents.*

In this same section of the preamble (column 1, 80 Fed. Reg. 3389), the Age ncy makes the following statement:

> *"The Agency is clarifying that if the preauthorization plan is approved in advance for chemical or biological agent use under specified discharge circumstances, then the OSC may authorize the use of the agents on the Sc hedule for their intended purpose without the incident specific concurrences and consultations described in paragraphs (b) of this section unless otherwise directed by the Administrator in accordance with current concurrence authority."*

We believe that the preauthorization plan should provide the OSC sufficient authority to use chemical or biological response agents. Since the Agency will have already been involved in the preauthorization plan approval, additional Administrator concurrence is unnecessary and would unduly complicate the response.

D. *RRTs and A rea Committees should be required to address and document their review and conclusions on whether or not the use of chemical or biological agents is appropriate.*

The Proposed Rule reads that:

> *"RRTs and Area Committees shall address in a preauthorization plan, as part of their planning activities, whether the use of chemical and biological agents listed on the Schedule on certain oil discharges is appropriate."*

This appears to compel RRTs and A rea Committees to document when and why such agents are appropriate, but does not require documentation or the rationale if they are not deemed appropriate, leaving the option open for RRTs and A rea Committees to simply not address prea uthorization for use of such agents. RRTs and A rea Committees should be required to address and document their review and conclusions on whether or not the use of chemical or biological agents is appropriate. API and NOIA suggest modifying the proposed language to read:

RRTs and A rea Committees shall address in a preauthorization plan, as part of their planning activities, whether or not the use of chemical and biological agents listed on the Schedule on certain oil discharges is appropriate and document the rationale for whether or not the use of an agent is considered appropriate.

IV. Section § 300.910(a)(1) Preauthorization Plan Development

A. *Dispersant use should not be limited by the preauthorization plan.*

EPA should not unnecessarily limit the use of dispersants through new req uirements in developing preauthorization plans. In this section (column 3 , 80 Fed. Reg. 3422), the Agency makes the following statement:

> *"For discharge situations identified where such agents may be used, the preauthorization plan must specify limits for the quantities and the duration of use* (Ed. emphasis added), *and use parameters for water depth, distance to shoreline, and proximity to populated areas."*

We assume that this provision in the Proposed Rule is a direct response to recommendation C5 on pag e 270 of the January 2011 report *Deep Water: the Gulf Oil Disaster and the Future of Offshore Drilling* by the National Commission on the BP *Deepwater Horizon* Oil Spill and Offshore Drilling. However, in the four years since the publication of this report, there have been numerous scientific, peer -reviewed environmental and health publications that have examined the dispersant use during DWH.[29] Subsequent laboratory toxicological studies of dispersed oil in the water column after dispersant application showed that there was no detectable harm to the species tested. Key findings from this research program included the following:

- Following surface dispersant applications, the dispersed oil c oncentrations in the water column return within 3 hours to background l evels due to vertical and horizontal dilution into surrounding ocean waters;

[29] Atlas, R. M., & Hazen, T. C. (2011). Oil biodegradation and bioremediation: A tale of the two wo rst spill in U.S. history. *Environmental Science & Technology, 45* (16), 6709-6715. doi:10.1021/es2013227; Chakraborty, R., Borglin, S. E., Dubinsky, E. A., Anderson, G. L., & Hazen, T. C. (2012). Microbial response to the MC -252 oil and Corexit 9500 in the Gulf of Mexico. Frontiers in Microbiology, Ecotoxicology and Bioremediation, 3, 357 -367. doi:10.3389/fmicb.2012.00357; Lubchenco, J. (2010, December 29). Oil spill clarifies road map for sea turtle recovery. The Miami Herald. Retrieved from http://www.nmfs.noaa.gov/mediacenter/docs/2011/mar/oil_spill_clarifies_road_map_for_sea_turtle_recovery.pdf ; Louisiana Department of Wildlife and F isheries. (2010, July). Oyster stock assessment report of the public oyster areas in Louisiana. Oyster Data Report Series No. 16. Retrieved from https://www.wlf.louisiana.gov/sites/default/files/pdf/page_fishing/32695 -Oyster%20Program/2010-oyster-stock-assessment-report.pdf; Operational Science Advisory Team. (2010). Summary report for sub-sea and sub -surface oil and dispersant detection: Sampling and monitoring. Retrieved from http://www.restorethegulf.gov/sites/default/files/documents/pdf/OSAT_Report_FINAL_1 7DEC.pdf; Operational Science Advisory Team. (2011). Summary report for sub -sea and sub -surface oil and dispersant detection: ecotoxicity addendum. Retrieved from http://www.restorethegulf.gov/sites/default/files/u306/FINAL%20OSAT%20Ecotox%20Addendum.pdf ; Shea, D. (2014, August 15). In re oil spill by the oil rig "Deepwater Horizon" in the Gulf of Mexico, on April 20, 2010. Expert report of Dr. Damia n Shea. New Orleans, LA: United States District Court, Eastern District of Louisiana. Retrieved from http://mdl2179trialdocs.com/releases/release20150120070000/TREX -013444.pdf; Soniat, T. M., King, S. M., Tarr, M. A., & Thorne, M. A. (2011). Chemical and physiological measures of oysters (Crassostrea virginica) from oil -exposed sites in Louisiana. Journal of Shellfish Research, 30(3), 713 -717. doi:10.2983/035.030.03 11; and U.S. Food and Drug Administration. (2010, October 29). NOAA and FDA announce chemical test for dispersant in Gulf seafood. Retrieved from http://www.fda.gov/NewsEvents/Newsroom/PressAnnouncements/2010/ucm231653.htm.

- The surface dispersants were applied over an operating area of 18,000 sq. mi., as a result, surface dispersant application rarely occurred in the exact same location, which further enhanced dilution of dispersed oil in the Gulf; and
- The mixing of relatively small volumes of dispersed oil into the very large, well -mixed Gulf of Mexico resulted in very short exposure durations of animals

These studies, conducted during and after the DWH aerial dispersant operations, indicate that there is no need to specify limits for the quantities and the duration of use of dispersants in a preauthorization plan. We believe the Section 300.910(a)(1) proposed requirement that specifies limits for the quantities and duration of use should be deleted from the Proposed Rule.

If the Agency wants to determine whether and how to limit dispersant use, we propose a joint EPA/industry study of existing research results to provide scientific credibility to any limitations that are established. Without scientific justification, artificially -set limitations on dispersant use would threaten the intent of using NEBA to support response decision -making and would limit the OSC's ability to make the best overall decision concerning the actual scenario and the environment.

To be clear, API and NOIA believe the amount of dispersant used should be based on the outflow of oil and the dispersant-to-oil ratio (DOR) that is effective in dispersing the plume. It is under stood that after a specified time period (e.g., 48 hours), the OSC will request RRT concurrence for continued dispersant use (except when there is a threat to human life and health) wherein concerns over quantities and duration can be evaluated.

We believe that the proposed preauthorization requirement to provide distance to shoreline and proximity to populated areas is not justified as the RRTs/Area Committees have already set these requirements in existing preauthorization plans as appropriate . Each RRT has developed preauthorization maps which clearly depict minimum distances from shore and minimum water depths. In addition to these already established distance and depth limitations, during an actual response, dispersant responders work with the RRT, OSC, and Responsible Party to set standard requirements and operating procedures for aerial spray safety setbacks of 1,000 ft. or more from vessels, platforms, response personnel, and sensitive marine life. Because these practices are already in place, there is no need to specifically regulate new parameters. For these reasons, API and NOIA believe the parameters for distance to shoreline and proximity to populated areas should be deleted.

We suggest that the Agency confer with other Federal agencies involv ed in oil spill planning and response and revise the language in the Proposed Rule to specify that preauthorization should rely on the concepts of CERA/NEBA as the foundation for pl ans and response decisions, not on artificially set limits on dispersant volumes. The USCG has conducted more than a dozen CERAs to help inform the decision-making process, but the Proposed Rule appears to contradict the CERA/NEBA tenets that were previously used to define preauthorization zones.

B. *EPA should work with other agenc ies to implement changes to preauthorization plans.*

Finally, since the Proposed Rule suggests that the Agency is adjusting the natural resource trustee concurrence requiremen t for preauthorization plans, we believe strongly that the Agency must demonstrate how it will work with other agencies to implement these proposed changes into the existing RRT preauthorization process and should specify a time interval for the relevant agencies to complete this integration (e.g., 1-year).

C. *The USCG and Bureau of Sa fety and Environmental Enforcement (BSEE) already regulate logistical aspects of dispersant use.*

There is no reason for the Agency to regulate logistical matters related to dispersant usage because they are already regulated by other agencies. In Section § 300.910(a)(1), the Agency makes the following statement:

> *"...and logistical factors including inventory, storage locations and manufacturing capability of available agents, availability of equipment needed for agent use, availability of adequately trained operators, and means to monitor agent use in the environment."*

There have been many advances in regulations and spill response planning in the past decad e. The regulations should acknowledge that there are now requirements in 33 CFR Parts 154 and 155 that have established a surface dispersant capability throughout the U.S. to apply up to 55,000 gallons (daily) of dispersant out to 50 nm offshore. These new regulations resulted in the development of 44 strategically located dispersant stockpile bases. Fur ther, these regulations require training and exercising of equipment and personnel. Given existing compliance with 33 CFR Parts 154 and 155, preauthorization plans do not have to restate all of these assets but should be able to refer to this regulated cap ability. With current U.S. stockpiles containing close to 1 million gallons of dispersant and international stockpiles containing over 1.5 million gallons (as established via the Global Dispersant Stockpile), the requirement to assess dispersant manufacturing capability during the initial response is unnecessary.

We recommend that the Agency revise this section of the regulation to not require specific information on stockpiles, spray equipment, personnel, training or manufacturing and acknowledge that the USCG and the BSEE are regulating these aspects of spill response.

V. <u>Section § 300.910(a)(2) – Preauthorization Plan Approval</u>

A. *The proposed language for w hen a withdrawal of concurrence invalidates a Preauthorization Plan is unclear.*

EPA needs to clarify when withdrawal of concurrence invalidates a preauthorization plan. In the preamble to this section (column 2, 80 Fed. Reg. 3389), the Agency states:

> *"The Agency is proposing specific procedures for concurrence withdrawals, allowing agencies to do so if they believe the preauthorization plan no longer addressed or reflects existing situations if it were to be implemented. While an agency with concurrence authority may not decide to withdraw concurrence from an approved preauthorization plan, there are curr ently no set procedures to promptly address those situations. The proposal would require the RRT and the ACs to address the withdrawal of approval of the preauthorization plan within 30 days of the withdrawal, allowing an opportunity to address the concerns."*

However, the Proposed Rule language is not clear regarding how the withdrawal process operates. While the proposed 30-day period for the RRT to address withdrawal is a positive step as it provides structure and guidance to the process, it is unclear w hen withdrawal of conc urrence would render the preauthorization plan invalid. API and NOIA believe this only makes sense if there is no active or ongoing response. Agencies should not withdraw their concurrence from an existing plan during a response as this could make responding in a timely fashion difficult, if not impossible. This could potentially allow an agency to invalidate a plan at the time of a s pill forcing a delay in the use of dispersants at the most critical time of the response. Note that the States of Alaska and Florida have withdrawn their approval for

preauthorization in state waters, which was done effectively without any additional regul atory requirements.

Clarifying the language to allow for preauthorization plans to remain in place dur ing the 30-day period in which RRTs and A rea Committees address the withdrawal would prevent disruption to response activities. This would maintain the validity and usefulness of preplanning and preauthorization, while allowing for agencies to withdraw con currence based on changing conditions, reducing the possibility of the withdrawal based on political or media influence. This 30 -day review period should also be subject to public comment.

We also note that currently Subpart J does not include the USCG representative to the RRT as a government agency required to approve preauthorization plans. It is important to inc lude one of the key stakeholders, the USCG, in the approval of preauthorization plans.

Accordingly, API and NOIA suggest modifying the proposed language as follows:

> *(2) Preauthorization Plan Approval. The EPA representative to the RRT, USCG RRT representative for the coastal a nd ocean zones where the USCG OSC would have primary authority and responsibility for operational decisions, the Department of Commerce and the Department of the Interior natural resource trustees and, as appropriate, the state RRT representatives shall review and either approve, approve with modification, or disapprove the preauthorization plans developed by the RRT and/or the Area Committees. Withdrawal of concurrence means the preauthorization plan becomes invalid and the authorization of use for chemical or biological agents must be performed according to paragraph (b) of this section. For an agency to withdraw concurren ce, the agency must first notify the RRTs and Area Committees. The RRTs and Area Committees shall review and address the withdrawal within 30 days of the date of notification, during which time the notification of withdrawal will be subject to public comment. The Preauthorization Plan will remain in effect while the RRTs and Area Committees review the withdrawal. If the agency withdrawing its concurrence remains unsatisfied after the 30 -day review period, then the Preauthorization Plan becomes invalid at th e conclusion of the 30 -day review period, and the authorization of use for chemical or biological agents must be performed according to paragraph (b) of this section.*

VI. Section § 300.910(a)(3) – Preauthorization Plans Reviews

A. *The re-approval process for pr eauthorization plans is too frequent and the timing and process for review and revision is unclear.*

Currently there are no requirements for the review of preauthorization plans. Revised Section 300.918(a)(3) requires preauthorization plan review, and revision as needed at the following times:

> *at least every 5 years; after a major discharge or a Spill of National Significance (SONS); to address revisions of the Schedule; to reflect new listings of threatened and/or endangered species; and to address any o ther change that may impact the conditions under which the use of chemical and biological agents is preauthorized.*

The preamble explains that "other changes" may include a new or revised worst case discharge estimate, and that this review requirement is "intended to ensure that preauthorization plans are actively maintained and updated to reflect revisions to the Schedule." (80 Fed. Reg. 3389). If the Agency decides to move forward with a re view requirement, API and NOIA recommend that the Agency clarify (1) what defines a

"major discharge"; (2) what process should be followed for re-approval; and (3) what effect a change in the preauthorization would have on existing FRPs and VRPs, which are also updated on a 5 -year cycle. We believe the review of preauthorization plans every 10 years or, at the discretion of the RRT/ Area Committee after a SONS (per [a] and [b] in the statement) , is sufficient and that the "major discharge" language should be removed . The Agency should also strike the requirement for review upon new threatened and/or endangered species listings and "to address any other change" since this type of planning on a "less than five year basis" overly -burdens the RRTs. Also, the Agency should clarify what the re-approval process would entail and establish a framework for how long review and approval should take. In addition, the USCG representative to the applicable RRT should also be included in this process. This, along with the remaining review triggers, would still allow for robust, current preauthorization plans, while maintaining stability of existing, potentially affected FRPs and VRPs.

We suggest modifying the proposed language as follows:

> *(3) Preauthorization Plans Reviews. The RRT and/or the Area Committees must review, and revise as needed, pursuant to paragraph (a)(1) of this section, preauthorization plans at least every 10 years; or at the discretion of the RRT and/or Area Committee after a Spill of National Significance (SONS); to address revisions of the Schedule; to reflect new listings of threatened and/or endangered species; and to address any other change that may impact the conditions under which the use of chemical and biological agents is preauthorized. The designated EPA RRT representative, the USCG RRT representative for the coastal and oceans zones where the USCG OSC would have primary authority and responsibility for operational decisions, the Department of Commerce and the Department of the Interior natural resource trustees, and the RRT representative from the state(s) with jurisdiction over the waters of the area to which a preauthorization plan applies shall review and either approve, approve with modification, or disapprove any revisions to the preauthorization plans.*

VII. Section § 300.910(b) Use of Agents Identified on the Schedule on Oil Discharges Not Addressed by a Preauthorization Plan

A. *A reasonable consultation time of 48 hours should be established for natural resource trustees if the Agency concludes that the current "if Practicable" standards should be changed*

In this section in the Proposed Rule (column 1 , 80 Fed. Reg. 3423), the Agency has removed the words *"when practicable"* with respect to consultation with the Department of Commerce (DOC) and the Department of the Interior (DOI) natural resource trustees for the following reasons discussed in the preamble:

> *"The requirement for consultation with the DOC and DOI natural resource trustees is also maintained. However, the language is amended by removing 'when practicable' with respect to consultation with the DOC and DOI natural resource trustees. The Agency believes that the case-by-case decision making should include consultations with natural resource trustees since these discharge situations may present unique challenges when selecting a response option that involves chemical or biological agents. While the Agency recognizes the time -critical nature of decision making during a response, advances in communication technology with increased capabilities to communicate quickly."*

(column 1, 80 Fed. Reg. 3390) API and NOIA believe that while technical advances in communication (e.g., cell phones) have improved since the last Subpart J revision, it does not guarantee an immediate or timely response by the required individuals. And the current system has worked well for years. Thus,

continued flexibility in the approval process (i.e., "when practicable") is still warranted to ensure the timeliest and best possible response . If the Agency concludes that the current " when practicable" language needs tightening up it could establish a reasonable consultation time of 36 hours. Spill response decision-making requires timely decision-making. Natural resource trustees already have an independent process to assess possible damages outside of the response effort (via Natur al Resources Damage Assessments [NRDA]). The phrase could be revised to state, "*...when practicable within a 36-hour timeframe*".

B. *The change in th e term "navigable waters threatened " will set up additional barriers to dispersant use and unduly limit actions by the OSC.*

In the same paragraph (column 1, 80 Fed. Reg. 3390), the Agency makes the following statement:

> *"The Agency is also proposing to revise the term 'navigable waters threatened' to 'waters and adjoining shorelines threatened' to be consistent with the provisions in paragraph (a) of this section."*

We are concerned that this change will set up additional barriers to dispersant use and unduly limit actions by the OSC.

C. *Early dispersant use decision-making should focus on documenting that a NEBA justification exists to support dispersant application.*

The Proposed Rule would unnecessarily establish new requirements f or an OSC to document the parameters used to authorize dispersants. This is discussed in the same section (column 2, 80 Fed. Reg. 3423), as follows:

> *"...the OSC must consider and document the parameters for the use of agents including the quantities to be used, the duration of use, the depth of water, the distance to shoreline and proximity to populated areas, and should address factors suc h as environmentally sensitive resources or restricted areas that might be impacted, agent inventory and storage locations, agent manufacturing capability, availability of equipment needed for agent use, availability of adequately trained operators and appropriate means to monitor agent use in the environment."*

Because dispersants have a limited window of opportunity to be used on spills, it is important to decide quickly on their use. Decisions by the OSC should not be delayed to complete information that is not needed. Many of the proposed items for the OSC to document (e.g., stockpiles, personnel, spray assets) are already specified and required in the applicable response plan requirements of 33 CFR Parts 154 and 155. Given the current stockpile availability in the U.S. and in the Global Dispersant Stockpile, dispersant availability will not be a concern for weeks or months. As suc h, we believe that the immediate actions by the OSC should include a review of the pre -authorization plan(s) and a rapid , incident-specific NEBA to support the selection of response options that may include dispersant application as a means to lessen or eliminate impacts to sensitive shoreline resources or to pr otect worker health and safety. In addition, we recommend that the Pr oposed Rule require referencing the USCG -classified dispersant OSRO or other OSRO for many of the items requested.

VIII. <u>Section § 300.910(c) Burning Agents</u>

A. *Ignition devices for in-situ burning are not burning agents.*

Under the current rule an OSC may author ize use of a burning agent on a case -by-case basis after concurrence of the EPA representative to the RRT, the state RRT representative and in consultation with the DOC and DOI natural resource trustees, when practicable. The Agency proposes to authorize the OSC to approve the use of a burning agent without the approval or consultation with other agencies to facilitate expedited use in recognition of their increasing importance in response to spills and that small quantities of burning agents are used and are quickly consumed by the resulting fire. API and NOIA agree and commend the Agency for this proposal. Further, we agree that ignition devices should be excluded from the definition of burning agents.

We believe that chemical herder agents are includ ed within the scope of the term burning agents. In the preamble to this section (column 3, 80 Fed. Reg. 3390), the Agency makes the following statement:

> *"Further, because of the nature of burning agents and the proposed revisions to the authorization of use for these products, the Agency continues to believe it is not necessary to require product submissions for burning agents."*

We believe this statement also applies to chemical herder agents. Please clarify if this is correct.

IX. Section § 300.910(d) Exception

A. *EPA needs to clarify the phrases "protection of human life" and "near inhabited areas."*

In this section in the Proposed Rule (column 2 , 80 Fed. Reg. 3423), the Agency makes reference to "the protection of human life ." We believe it is unclear whet her "*protection of human life* " is inclusive of the protection of worker health and safety. The Agency should amend this section to clearly include the protection of oil spill response worker safety within this concept. This same section also references spi lls "*near inhabited areas.* " This language is confusing since spills where dispersant use serves an environmental benefit will rarely (if ever) be near inhabited areas.

B. *A time limit on protecting human health is inappropriate.*

We believe the establishmen t of a 48 h our limit to the OSC's use of authority to use agents to protect human health is inappropriate. There should never be a limit to the OSC's ability to make decisions that are aimed at protec ting human health or human life, and the exception shoul d be permissible until the threat to human health has subsided. The current rule adequately allows for the OSC to approve use of dispersants in such cases and EPA has not justified that the proposed requirement is necessary or provides any benefit to existing requirements.

X. Section § 300.910(e) Prohibited Agents

A. *Banning nonylphenol (NP) and nonylphenol ethoxylates (NPE) is not necessary for dispersants.*

API and NOIA submit that NP and NPE components should not be banne d. In the preamble to this section (column 2, 80 Fed. Reg. 3391), the Agency makes the following statement:

> *"The Agency is also proposing to add a prohibition from listing on the Schedule and from authorizing use of any chemical or biological agents that contain n onylphenol (NP) or nonylphenol ethoxylates (NPEs) as components."*

While we appreciate the Agency's concern about NP and NPE components, an outright prohibition seems inconsistent with the practice of allowing NP and NPE ingredients in a variety of widely-used cleaning products in the U.S. After use, these products are routinely flushed and transported into wastewater discharges on a daily basis. Spill response dispersant use is extremely infrequent and only occurs when a significant threat to shoreline or near shore resources is present. Banning these components in oil spill dispersants while allowing their wide spread use in other commercial products seems contradictory and prejudicial. API and NOIA requests that the Agency explain why NP/NPE is banned in dispersant products, but not in many more commonly manufactured products.

B. *Endocrine disruption testing has little relevance and is not necessary.*

We submit that endocrine disruption testing should not be added. In the same section (column 1, 80 Fed. Reg. 3392), the Agency states:

> *"Because validated testing methods are still being developed the agency is not proposing any test requirements with respect to endocrine disruption for products to be listed on the Schedule."*

We believe that adding any such testing provides little relevance in the context of oil spill response decisions and that testing efforts are better directed at the use of standardized toxicity tests with aquatic organisms using endpoints of population relevance (i.e., mysid and *Menidia* tests proposed).

C. *Mechanical response clean-up methods have greater limitations in offshore conditions than dispersants.*

Mechanical response clean-up methods have many limitations. In the same section (column 2, 80 Fed. Reg. 3392), the Agency makes the following statement:

> *"There are chemical and biological agent alternatives to sinking agents and to agents containing NP or NPE; <u>as well as mechanical methods</u>* (Ed. emphasis added) *for responding to oil discharges, including those situations that pose extreme threats or are time critical."*

We believe the Agency must realize the limitations that mechanical response methods have in offshore conditions. Weather conditions will preclude booming and skimming operations in sea states above six feet wave height, and even in ideal sea state conditions, responding to offshore spills historically has resulted in 5-10% of spilled oil recovered by mechanical recovery methods. For this reason, it is critical that responders have reasonable access to chemical response methods, because of the substantially higher encounter rates (80% or more) that can be achieved by treating the spilled oil with dispersants.

XI. <u>Section § 300.910(f) Storage and Use of Agents</u>

A. *No additional requirements should be put in place for product shelf life.*

Currently, the manufacturer is required to provide information on recommended conditions of storage and use for each product at the time an application for listing a product under Subpart J is submitted to the Agency. With regard to shelf life, the Agency is requesting comments on whether more information should be submitted on issues related to shelf life. Specifically, in the preamble to this section (column 1, 80 Fed. Reg. 3393), the Agency makes the following statement:

> *"To alleviate concerns that applications are submitted that establish an extended or indefinite shelf life for a product, the Agency is requesting comments on whether any*

> *additional data or information requirements should be included for the product listing determinations specific to a product's shelf li fe, or whether alternative approaches, such as limiting the shelf life for product categories to a given timeframe, should be considered."*

API and NOIA believe that no additional requirements should be put in place for product shelf life and thus it shoul d not change the current practice of having the manufacturer provide information on recommended conditions of storage and use for each product. Ensuring product efficacy (regardless of age) is the only relevant end-point. For example, research has shown that properly maintained stockpiles of Corexit 9500 have remained effective for 25 years. Given that periodic effectiveness testing is proposed, there is no justification for also mandating a shelf life that could potentially result in costly replacement of stockpiles that remain viable and effective.

If the Rule dictates a dispersant shelf -life, this could caus e an undue financial burden. We believe that suppliers may set low shelf-lives to protect themselves from liability or to promote future recurring s ales, even though there are well-documented studies indicating that properly stored dispersant stockpiles have maintained their effectiveness for decades. A 2008 study by Clark, *et al.*,[30] evaluated U.S. stockpiles of Corexit 9527 and concluded that batches of this dispersant have remained viable for spill response for over 30 years. Establishing shelf -lives for products makes sense if periodic testing is not specified. However, since the Proposed Rule specifies re -testing of the dispersant stockpiles, there is no justification for also requiring a shelf -life. We submit that the requirement for a shelf -life should be removed from the Proposed Rule and that the Agency should instead provide more clarity on product re - testing for efficacy. Specifically, the requ irement to conduct periodic 5 -year tests of effectiveness and toxicity is unclear. The sampling requirement has not been delineated in the Proposed Rule, which could be interpreted, in the extreme case, that every single dispersant container would have to be re-sampled and tested to remain in an "approved" status.

The Rule should clearly state that representative samples of batches/lots are tested for efficacy and that the testing interval take into account the aforementioned studies that demonstrated tha t dispersant stockpiles maintain effectiveness for decades. Further, we submit that there is no scientific justification for re-testing toxicity.

B. *Central dispersant stockpiles are typically maintained by OSROs, not by Responsible Parties or product manufacturers.*

Under the Proposed Rule the OSC would only be able to authorize use if the Responsible Party certifies that the products have been stored under the conditions provided by the submitter. In this section in the Proposed Rule (column 2-3, 80 Fed. Reg. 3423), the Agency makes the following statement:

> *"The OSC may authorize for use only products that are certified by the Responsible Party to have been stored under the conditions provided by the submitter under 300.915(a)(6) and whose date of use does not exceed the expiration date listed on the container's label at the time of the incident. The Responsible Party must provide the OSC product documentation, developed in consultation with the submitter of the product to the Schedule, prior to OSC authori zation of product use affirming it has maintained its integrity, including no changes in its composition, efficacy, and toxicity."*

[30] Clark, J., Becker, K., & Lessard, R. (2008). Maintaining dispersant stockpiles and assessing their quality. *Proceedings of the 2008 International Oil Spill Conference, 2008*(1), 695-698. doi:10.7901/2169-3358-2008-1-695

The Agency needs to recognize that central dispersant stockpiles are typically maintained by OSROs and are stored mainly in 200 to 300 gallon totes so that they can be transported by truck or air to the site of a spill. In most spill situations, the dispersant stockpile will not be in the control of the Responsible Party or the submitter of the product (i.e., the original manufacturer) at the onset of a spill. As a result, there is no way for these two entities to make a certification for a product that was not in its direct control just prior to the spill. API and NOIA suggest modifying the language to clarify that Responsible Parties must only demonstrate compliance with Section 300.915(a)(6) for inventory/product they control.

C. *Certification of storage conditions should not be required for authorization of use.*

Section § 300.910(f) states that the OSC may authorize for use only products that have been stored under the conditions specified by the manufacturer including the maximum, minimum and optimum temperatures as well as humidity and other relevant conditions. This proposed regulation could require dispersants to be stored in climate controlled buildings with continuous temperature and humidity monitoring. We do not support this proposed requirement.

D. *API and NOIA opposes any re-testing requirements during a spill response.*

The intent of this section is difficult to discern but it appears that the certification requirement appears to essentially requires complete retesting of the product (at the time of the spill) for the Responsible Party to certify such a statement. It appears that the Agency has included this language with the specific intent to delay the dispersant authorization process by requiring complete re-testing at the time of a spill incident. We strongly oppose any re-testing requirements during a spill response. Any product testing should be conducted at a specified, planned interval, not during an emergency response. During a response, the Responsible Party should not have to certify the dispersants that will be applied, since that is the purpose of the NCP Product List. The only requirement should be for the Responsible Party to show the OSC, if requested, the original manufacturer's verification (Certification of Analysis) that the product meets specifications for dispersants with an age of less than the period for retesting recommended by the manufacturer. For older dispersants the Responsible Party should, if requested, show the efficacy re-test data for the stockpile. The test required should only be for effectiveness as it is a reliable indicator that the composition has remained the same. We propose that this language is changed to read: *"The OSC, to authorize the use of dispersants, may request that the Responsible Party provide test data of the dispersant stockpile showing it is effective."* We also suggest modifying the language so that it applies only for product manufactured or produced after the date of final rule publication.

E. *Re-testing requirements must permit flexibility in test results over time.*

This same section specifies re-testing and re-labelling requirements. This section does not specify the criteria for re-testing results and needs to specify permissible levels of deviation from past testing results. We point out that efficacy and toxicity testing have inherent variability. Two testing runs of the same product, in the same laboratory, in the same week can yield varied results, so any re-testing requirements must permit flexibility in test results over time. We suggest an allowable 10% variance in effectiveness test results, based on manufacturer and scientific experts' opinion on effectiveness testing. As previously stated, we do not believe there is any scientific justification to re-test for toxicity since efficacy results will confirm if the composition of the dispersant has changed. We suggest clarifying language so that it is clear product testing is only required if it is not stored under requirements provided by submitter, or it has expired.

F. *The assignment of an arbitrary 5-year life to products is biased and creates unwarranted expense due to recurrent testing.*

Finally, the assignment of an arbitrary 5-year life to products without regard to prior research on product viability over time, or on manufacturer input on product re-testing, is biased and creates unwarranted expense due to recurrent testing. Proposing 'no change' in composition, efficacy, and toxicity seems unreasonable and could result in significant costs of replacing usable dispersant stockpiles. Additionally, dispersant manufacturers typically advise that container seals not be broken and the material not be exposed to air, which would be necessary to test existing stocks, which may or may not have an expiration date printed on the label under current regulation.

Studies exist that show Corexit products maintain effectiveness for 25 years. We believe that any re-certification term should be based on existing scientific and manufacturer data. There should also be an allowance for some change/degradation of product over time, aligned to international practice. For example, the United Kingdom Marine Management Organisation allows a loss of efficacy to 75% of the original minimum level. Further, there is no scientific rationale to require any form of toxicity re-testing of a dispersant that was previously tested. Manufacturers will have the best insight regarding when product recertification should be needed. In addition, the Responsible Party should never be required to re-label products they did not produce, with the possible exception of adding a re-test date for product efficacy.

XII. Section § 300.910(g) Supplemental Testing, Monitoring, and Information

A. *API believes the standard efficacy and toxicity tests already required are more than adequate for regional use.*

We do not believe there is a need for additional testing than is currently required under the NCP. In the preamble to this section (column 2, 80 Fed. Reg. 3393), the Agency makes the following statement:

> *"The proposal removes 'in addition to the test methods specified in 300.915 and described in appendix C to part 300.' While RRT's may want to use the efficacy and toxicity testing protocols specified in Appendix C of the NCP for comparative purposes, the proposed revisions clearly establish the RRT's authority to require tests using parameters beyond those specified in Appendix C. Furthermore, there may be supplementary toxicity and efficacy testing information based on recognized standard testing methods already available that RRTs may want to consider when addressing site, area, or ecosystem specific concerns."*

We believe the standard efficacy and toxicity tests already required are more than adequate for regional use. We suggest the Agency consult the NOAA Dtox database. A review of that database suggests that the sensitivities of all tested species are in the known range and that additional tests simply reinforce the existing species sensitivity distribution curves. Regional data are unlikely to fundamentally change overall NEBA considerations for dispersant use. Any regionally required laboratory efficacy tests during a response will not be as relevant as a pilot test in the field and will only delay the dispersant authorization process approval beyond the window during which dispersants could be applied (thereby precluding their use in some situations where they may have provided a net environmental benefit). This proposal allows RRTs to request a broad range of additional data on a product, a situation that creates the potential for confusing and costly testing and is unlikely to add further information of value to oil spill response decision-making.

B. *Additional monitoring during a response is unreasonable.*

Section § 300.910(g) provides the RRT with the authority to request that the OSC require the Responsible Party to conduct additional monitoring associated with use of a product. We submit it is unreasonable to require additional monitoring during a response except in exceptional circumstances. In the preamble (column 2, 80 Fed. Reg. 3393), the Agency makes the following statement:

> *"For example, the RRT may want to monitor the exposure of marine mammals to oil constituents, including the dispersed oil, or to monitor toxicity in the water column using biological assays."*

We believe such monitoring during a response is unreasonable and should only be undertaken if (a) there is some critical question associated with the spill response that has not been answered with the testing already in place; and (b) there is sufficient time to arrange for proper and scientifically defensible protocols to be established. Given that tests associated with such monitoring are likely to be complex and require considerable time to prepare, implement, and interpret, it appears that the tests would not inform operational decision-making during the response, but rather would develop data for NRDA. As previously stated, operational monitoring and NRDA are two different things and this Proposed Rule should not specify NRDA monitoring. Additionally, the "biological assays" that are referenced in the Proposed Rule are impractical given the nature of offshore spills where dilution is significant, water movement is constant, and synoptic sampling of small quantities of organisms from dynamic water masses is not representative of any chronic or resident conditions.

API and NOIA suggest revising the language as follows:

> *(g) Supplemental Testing, Monitoring and Information. The RRT may require supplementary toxicity and efficacy testing, or available data or information that addresses site, area, or ecosystem specific concerns relative to the use of a product for both planning and authorization of use. During a discharge incident, the RRT may request that the OSC require a Responsible Party to conduct additional monitoring associated with the use of a product to the extent necessary for conducting response operations. Such additional monitoring data may include supplemental sampling and testing to ensure safety of response personnel, to assess acute environmental impacts or otherwise aid the OSC and/or the RRT in operational decisions.*

XIII. Section § 300.910(h) Recovery of Agents from the Environment

A. *The OSC has the authority to use the NEBA process to make decisions on recovery of agents from the environment.*

In the preamble to this section, the Agency is requesting comments on new language that would require the Responsible Party, under OSC oversight, to remove specified products/agents from the environment after use. While the Proposed Rule hints at NEBA, it does not reference it specifically when suggesting that some situations (e.g., safety of personnel or greater harm to the environment) might preclude the removal of these agents. That would seem to contradict the main change in this section (h) while not providing any framework (e.g., NEBA) in which to make a determination for precluding the removal. The language is also potentially redundant as the OSC is already obligated to consider safety and additional adverse environmental impacts (i.e., via NEBA) as part of the normal decision-making process. Essentially, it is already an underlying tenet of the response process. Additionally, in some cases recovering 100 percent of these agents after application is not practical or even feasible regardless of the diligence of the Responsible Party

We suggest revising the proposed language to make it clear that the OSC has the authority to use the NEBA process to make these decisions:

> *(h) Recovery of Agents from the Environment. Depending on factors such as the safety of response personnel and harm to the environment (including consideration of Net Environmental Benefit Analysis [NEBA]), and as directed by the OSC, the Responsible*

Party shall, following application, endeavor to adequately contain, collect, store and dispose of the agents that are intended to be recovered from the environment.

XIV. Section § 300.910(i) Reporting of Agent Use

A. *The Agency should not adopt a prescriptive time period for reporting of agent use.*

The Agency is requesting comments on the appropriateness of requiring the OSC to provide the RRT additional reporting information " *within 30 days of completion of agent use* ." This runs counter to the adaptive decision-making that might be required during a discharge event where such agents are used.

Based on the pre authorization testing that is being proposed and, more importantly, the variability of discharge situations, we recommend that the Agency not adopt a prescriptive time period and rather provide the relevant RRT(s) discretion to determine the appropriate time period for reporting agent use. A strong justification for this approach is that during prolonged discharges, a fixed 30 -day reporting period might constrain the response personnel and interfere with higher priority resource allocations. Furthermore, the OSC would typically request this information as part of the daily briefing.

API and NOIA suggest removing the 30-day window for reporting and modifying the proposed language as follows.

> *(i) Reporting of Agent Use. Unless already included in the OSC report required under § 300.165 of this part, the authorizing OSC shall provide the RRT the following information on chemical and biological agents used in response to an oil discharge: product name, quantity and concentration used, duration of use, and locations.*

XV. Section § 300.913 Monitoring the Use of Dispersants

A. *The appropriate use of spill mitigating products will reduce t he overall impact of the spill and protect the health and safety of response personnel.*

The Agency makes the following two related statements to explain the goal of the NCP Product Schedule (column 3, 80 Fed. Reg. 3393; column 3, 80 Fed. Reg. 3402; respectively):

> "*The goal of establishing a Schedule under the NCP is to protect the environment from possible damage related to spill mitigating products used in response to oil discharges.*"
>
> *"Emergency response personnel need to know whether a dispersant or an y other type of chemical or biological agent on the Schedule could have negative environmental impacts relative to the oil before decisions are made about its use in a particular oil discharge situation. Consequently, it is essential to consider comparativ e information about the efficacy and the toxicity of these products.*"

Possible damage related to spill mitigating products used in response to oil discharges is considered, but not the possible damage that could be caused by the oil itself without the us e of spill mitigating products. The main threat to response personnel, communities, and the ecosystem comes from the oil, not from the spill mitigating products. The concept of CERA/NEBA accepts that some effects might be caused by the response (including the use of spill mitigating products), but also recognizes the greater benefit of reducing the overall impact of the spill that can be achieved by using them. The Agency needs to acknowledge the importance of NEBA in justifying the use of dispersants.

We believe that the focus on the potential for negative environmental impacts that could be caused by a dispersant ignores the essential point that dispersants are used to mitigate (minimize) the negative human health and environmental impacts that could be caused by the oil. The main threat to the ecosystem comes from the oil itself, not from the dispersant. The wording of the preamble to a final rule needs to:

- reflect the ultimate aim of dispersant use;
- clarify that the toxicity of dispersed oil comes from the oil (not the dispersant); and
- acknowledge and emphasize the importance of using a NEBA process in dispersant use decision-making.

We request that the Agency consider including the following two paragraphs (or equivalent language) in the preamble to the final rule to briefly explain dispersant use and the importance of NEBA in making dispersant use decisions.

> *"Dispersant Use and Net Environmental Benefit Analysis (NEBA)*
> *The aim of dispersant use is to protect response personnel from VOC exposure and to prevent oil from drifting ashore and potentially causing severe and long-term damage to coastal resources and communities, by dispersing the oil into the water column where it will be diluted rapidly to very low concentrations and the majority of the oil will subsequently be biodegraded. This common aim of dispersant use is the same, whether it is applied to oil on the sea surface or directly at the outflow source in the case of a subsea release. The potential risk of dispersant use is that, by dispersing the oil into the water column, there is an increased potential for toxic effects to be caused to marine organisms. The severity of the toxic effects that could be caused is related to the exposure of the marine organisms to the dispersed oil (or the partly water-soluble oil compounds) and exposure is a function of concentration of the oil in the water and the duration for which such exposure persists.*
>
> *The issue of the environmental trade-offs caused by dispersant use is addressed by Net Environmental Benefit Analysis (NEBA). NEBA is a process used to make choices about oil spill response techniques to be used to minimize the potential impacts of the oil on people and the environment. NEBA takes into account the likely effectiveness and feasibility of conducting alternative response techniques under the prevailing conditions. The main principle of NEBA is that an oil spill response technique (or combination of techniques) that is used should be that which minimizes potential impacts of the oil on the environment to the greatest extent and promotes the most rapid recovery and restoration of the affected area. In many cases of dispersant use, the potential risks to the offshore marine ecosystem are very low because dispersed oil concentrations are low and persist only briefly. There is benefit in using dispersants in offshore environments to prevent damage to coastal communities and sensitive near-shore resources where persistent stranded oil can cause longer-term environmental problems."*

The Proposed Rule also appears to focus more on the environmental effects during the response rather than the health and safety of response personnel. The Agency should make it clear that one of the primary objectives of characterizing the efficacy and understanding transport is to protect the health and safety of response personnel. That critical aspect is not mentioned in this section.

B. *The OSC should determine where, when, and how monitoring resources should be deployed.*

EPA should clarify the meaning of the phrase *"upon initiation and for the duration of subsurface dispersant use."* Later in this section of the preamble (column 2, 80 Fed. Reg. 3394), the Agency states the following:

> *"The proposal requires the responsible party to implement monitoring for any subsurface dispersant use in response to an oil discharge upon initiation and for the duration of subsurface dispersant use."*

The phrase *"upon initiation and for the duration of subsurface dispersant use"* can be misconstrued to mean that all monitoring required by the Proposed Rule must be performed at all times, when in fact the monitoring timing and operational parameters should be established by the OSC. Every response is unique, and it should be left to the OSC to determine where, when and how monitoring resources should be deployed to provide the best near real-time operational monitoring on the progress of the response.

C. *The Proposed Rule should be restructured to separate surface dispersant application monitoring from subsea dispersant injection monitoring.*

EPA should distinguish between surface and subsea dispersant application monitoring. In the first paragraph of this section in the Proposed Rule (column 1, 80 Fed. Reg. 3424), the Agency states:

> *"As directed by the OSC, the Responsible Party must monitor any subsurface use of dispersant in response to an oil discharge, surface use of a dispersant in response to oil discharges of more than 100,000 US gallons occurring within 24 hours, and surface use of dispersant for more than 96 hours in response to an oil discharge, and submit a Quality Assurance Project Plan for approval to the OSC covering the collection of all environmental data".*

We believe it is significant that monitoring is to be required *"as directed by the OSC."* It should be made clear that the OSC has discretion to authorize initial use of dispersants during a time-limited efficacy testing phase while evaluating the need for and extent of additional adaptive monitoring actions. The OSC is in the best position to determine when and what operational spill response monitoring is to occur. Experience has shown that the use of phased monitoring approaches, consistent with the USCG *et al.*'s *Special Monitoring of Applied Response Technologies* (SMART) protocols, is appropriate, and they are most responsive to changes in site-specific conditions and information requirements. While the requirement for the Quality Assurance Project Plan (QAPP) is constructive in helping to maintain effective data integrity and management, the rule should provide the OSC with discretion in how detailed the document needs to be to meet this requirement. This would provide the relevant incident decision-makers with practical information that can be consistently acquired, analyzed, and understood.

The Proposed Rule seems to inappropriately replace the currently used 3-tier SMART protocols for monitoring surface dispersant use with the more exhaustive monitoring approach proposed for subsea dispersant use. The USCG Strike Teams already have the responsibility for monitoring surface dispersant operations and possess the trained personnel and equipment to conduct these activities. Many RRTs require the immediate activation of USCG Strike Team monitoring for all dispersant operations, regardless of size or location. We believe it is beneficial to have the government perform the surface monitoring to enhance credibility with the public that the operations are being conducted correctly and are working as intended. If this practice is going to change, the Agency should explain why it is circumventing an established, well-vetted process performed by the USCG Strike Teams. The information in the Proposed Rule creates confusion between existing responsibilities of the USCG Strike Teams and those of the Responsible Party. We request that this confusion be reconciled between the Agency and the USCG, as the Responsible Party should not be required to provide duplicate monitoring for surface dispersant operations.

For these reason s, we believe this section of the Proposed Rule should be restru ctured to separate surface dispersant application monitoring from subsea dispersant injection monitoring. API and NOIA offer the following relevant points:

- The section on surface dispersant application monitoring should specify that it will be implemented with existing SMART protocols using USCG resources.
- The section describing subsea dispersant injection monitoring should specify it will be performed by the Responsible Party.
- In both cases, the Proposed Rule should clearly state that the mobilization o f monitoring capabilities will not delay initial dispersant applications. Monitoring requirements should not impede the use of chemical agents during a response and should be implemented as appropriate resources become available during the response.
- The c riteria of greater than 100,000 gallons or duration of 96 hours for the spill should be replaced with a single criterion that " *any enhanced monitoring beyond SMART shall commence within seven days.* " Our rationale for a deployment schedule of seven days for any enhanced monitoring (whether subsea or surface dispersant monitoring) ensures that scientifically - defensible monitoring can be performed, by ensuring:
  - The best scientific experts can be assembled and mobilized to the spill;
  - Once on scene, the science team can design a monitoring plan that can be integrated safely into ongoing response operations, and be worked into the Simultaneous Operations (SIMOPs) schedule;
  - The team can tailor a general monitoring strategy into a specific mission plan for the monitoring vessel that is available (there is no way to know what research vessel may be available, and each vessel will have unique characteristics for equipment deployment);
  - Available monitoring vessel(s) can be identified, mobilized to port, and properly out fitted with the scientific equipment needed to implement the mission plan;
  - An appropriate sampling strategy can be developed and approved for the specific incident, taking into account ongoing response activities that may impede access to sampling sites; and,
  - Appropriate safety considerations for the monitoring vessel, dispersant application or injection platforms, and rapidly-assembled teams can be properly reviewed.

D. *Effectiveness determination is not currently scientifically feasible.*

The language in this section of the Proposed Rule appears to indicate the Agency believes that comprehensive monitoring will permit the dispersant effectiveness at sea to be quantitatively determined. This requirement for effectiveness determination is not currently scien tifically feasible. Also, use of the word "*comprehensive*" should be avoided as it is subjective. Instead, it should be replaced with "*adaptive*" which indicates that a decision could be revisited as the d ischarge situation changes. We recommend replacing all instances of the word " *comprehensive monitoring*" with "*adaptive monitoring*" throughout the proposed section. The following language could be used:

> *The agency believes that an adaptive monitoring program which includes objective science and ocean transp ort modeling in certain discharge situations is necessary to determine the overall effectiveness of the dispersants and should transcend the initial dispersant application to include the extent, character, and transport of the dispersant and dispersed oil in the water column.*

With respect to <u>surface dispersant use monitoring</u>, we offer the following comments:

- Experience at the DWH and other incidents has shown that no currently available remote sensing technique, including airborne visible/infrared spectro meter (AVIRIS) and other multi- and hyper-spectral equipment, is capable of accurately quantifying the total amount of oil on the sea surface. It is also not possible to accurately determine the total amount of dispersed oil in the water column at any time because water sampling or ultra violet fluorometry (UVF) transects cannot be made at sufficient resolution in space (i.e., distance and depth) and cannot be made at sufficient resolution in time (i.e., simultaneously at all points in the water under an oi l slick). Localized measurements by UVF made before and after dispersant spraying can reveal the relative effect of dispersant use, but is not quantitative . A sharp increase in dispersed oil concentrations in the water, after dispersant use, is a good indi cation that the dispersant is working, but such measurements cannot produce an aggregate estimate of disper sant effectiveness. As such, we believe that the Agency should remove any requirement for quantitative reporting of efficacy, since it is not technically possible.
- A QAPP should not be required for surface dispersant operations. QAPP for approval by the OSC covers the collection of all environmental data to ensure and maximize its quality, objectivity, utility, and integrity. All these requirements wil l result in the need to generate a very large document that needs to be reviewed, verified and approved by OSC. This will lead to delays in dispersants decision-making, while adding little value to the goals of the response. A more useful approach may incl ude a daily Quality Control review of the SMART data to determine if the surface dispersant applications were effective that day, to support decisions for possible dispersant use the next day. A Quality Control form was developed during DWH for this purpos e. we believe the QAPP requirement should be deleted for surface dispersant operations and replaced with a simplified Quality Control form for SMART data.

With respect to <u>subsea dispersant injection monitoring</u>, we provide the following comments:

- The Age ncy does not address methods for visual determination of subsea dispersant effectiveness. During DWH, visual monitoring of the subsea dispersant injection was conducted in two ways to provide valuable, near real -time information to help inform response dec isions. These included:
  - Aerial observations of surfacing oil quantity and appearance (provided from observation aircraft overflights); and
  - Subsea observations of changes in plume appearance at the injection site (provided from ROV imagery).

  This form of monitoring for subsea dispersant injection should be recognized as an important step for initially evaluating the operation while more "enhanced scientific monitoring capabilities" are being assembled.
- API has developed subsea monitoring guidance (API Techn ical Report 1152 , *Industry Recommended Subsea Dispersant Monitoring Plan – Version 1.0*) and recommends the Agency consider this as the basis for their approach for operational monitoring. This guidance is publicly available at http://www.oilspillprevention.org/~/media/oil-spill-prevention/spillprevention/r-and-d/dispersants/api-1152-industry-recommended-subsea-dis.pdf.

XVI. Section § 300.913(a) Documenting and Monitoring the Oil Discharged and Dispersants Used Requirements

A. *The Responsible Party should not be required to document information at the beginning of a response that has already been determin ed in planning and preparedness activities.*

The current regulations found in 33 CFR Parts 154 and 155 require OSROs that use dispersants to maintain a certain level of spray assets, dispersant stockpile quantities, and logistical capabilities to

deliver stockpiles to staging bases, and to provide information on the application dosages they can apply. Since this is regulated elsewhere, reporting of these elements should not be required in the Proposed Rule and should therefore be deleted.

The Responsible Party should not be required to provide the *"rationale for dispersant product choice(s) including the results of any efficacy and toxicity tests specific to area or site conditions."* If the dispersant product is on the Product Schedule (which means that eff icacy and toxicity data have already been provided to the government), then it is, by definition, a viable option for the response.

Providing *"hourly application rates"* does not apply to aerial and vessel application since dosages are prepared in gallons per acre application and are based on the spray assets' application speed an d spray system swath width. We suggest that this statement be modified to read: *"hourly application rates (in the case of subsea dispersant injection)."*

The phrase " *recommended di spersant-to-oil ratio (DOR)* " should be modified to " *recommended <u>initial</u> dispersant-to-oil ratio (DOR).* " Every spill is different, and the DOR should be adjusted as additional monitoring, analytical data results, and/or oil release rate information becomes available. Additionally, the DOR may need to change as the oil weathers or other conditions change. The Agency should consider providing a clarifying sentence recognizing that oil characteristics and effective DORs may vary and thus dispersant application rates/volumes cannot be fixed to pre-determined values as this practice may affect negatively the response.

The Agency should refrain from requiring estimates of dispersant volumes needed to effectively respond to the spill at the onset of the response. In the case of a subsea oil release, it is impossible to know how much oil will be released and , therefore, impossible to estimate what volume of dispersants might be necessary. Further, as oil weathers, DORs for surface application will need to be adjust ed, and these changing weathering conditions cannot be fully known in advance. Given current dispersant stockpiles in the U.S. and in the Global Dispersant Stockpile, it should be assumed that sufficient dispersants will be available if their use is warranted by NEBA.

B. *Identifying and reporting associated volatile petroleum hydrocarbons will be impossible with changing environmental conditions, and is unnecessary for response decision-making.*

The requirement to *"...document any associated volatile petroleum hydrocarbons..."* would be very difficult to achieve, given that volatiles will fluctuate with sun exposure and changing wind conditions. The reason for this requirement is not explained; therefore, the Agency should specify what is intended by this requirement or delete it.

C. *The Agency should accept best available industry practice for dispersant monitoring.*

There is no need to restate the requirement for documenting the results of toxicity testing. In general, the list of characteristics is practical and do es not create an undue burden ; however, there is a significant assumption in the requirements (i.e., that the condition of the discharge point is such that it will enable the acquisition of these data at any given time). A reference to a best available industry practice, or any of the existing guidance on dispersant sampling/monitoring (e.g., API guidelines on subsurface dispersant monitoring, API TR 1152) would be more effective for gathering the information requested by the agency without imposing undue b urden on response personnel. Additionally, there is no technical reason to differentiate subsurface discharges from the rest of this paragraph. We suggest removing this differentiation and suggests modifying the proposed language as follows:

As directed b y the OSC, the Responsible Party must monitor any subsurface use of dispersant in response to an oil discharge, surface use of dispersant in response to oil discharges of more than 100,000 U.S. gallons occurring within 24 hours, and surface use of dispersa nt for more than 96 hours in response to an oil discharge, and submit a Quality Assurance Project Plan for approval to the OSC covering the collection of operational monitoring data. When these dispersant use conditions are met, and for the duration of dispersant operations, the Responsible Party shall:

(a) Document the characteristics of the source oil; best estimate of the oil discharge flow rate, periodically reevaluated as conditions dictate, including a description of the method, associated uncertainti es, and materials; dispersant(s) product used, rationale for dispersant product choice(s), recommended dispersant-to-oil ratio (DOR); and the application method and procedures, including a description of the equipment to be used, hourly application rates, capacities, and total amount of dispersant needed.

XVII. Section § 300.913(b) Sampling Requirements

A. *Daily water sampling and testing is burdensome.*

The requirement to conduct these required tests on water samples daily may prove overly burdensome and not necessary after the initial efficacy tests. We suggest modifying the Proposed Rule to ensure that the OSC has discretion in h ow frequently the samples must be acquired after the initial efficacy tests are conducted. Sampling on a daily basis, once efficacy is determined and agreed, is arbitrary and does not fit the timeline required to provide in-depth analysis of samples.

The collection of water column samples where the dispersed oil plume is located is not necessary and in many cases is not possible for aer ial dispersant application. For spills of up to 100,000 gallons, aerial dispersant assets can be activated and apply dispersants to the entire spill area during Day 1, before a monitoring team can arrive on-site to conduct water sampling. It was shown during the DWH event that to collect water samples, special operations had to be developed and coordinated so that the vessel monitoring team had sufficient time to obtain the water samples of the dispersed oil. This was due to the fact that most of the area u nder the aerial dispersant application returned to normal background levels within one hour following the application. It should also be understood that water samples will only determine the amount of oil in the sample, not the effectiveness of the dispers ant, as there is no way of determining the amount of oil that was treated. As stated previously, these sampling requirements do not provide information on dispersant effectiveness and can therefore not inform the operational response. Water sampling requir ements are more appropriate to NRDA and should be directed under NOAA jurisdiction.

B. *The oil droplet size distribution measurement, as described, is not technically feasible.*

The requirement in 300.913(b)(1) to conduct *"in situ oil droplet size distribut ion, including the mass or volume mean diameters between droplet sizes ranging from 2.5 to 2000 µm* " is not possible for three reasons:

- The General Information section suggests that this measurement can be done via a Laser In Situ Scattering and Transmissometry (LISST) instrument, but the LISST detection limit is roughly 500 microns;
- Commercial droplet size measurement equipment suitable for deep water monitoring that ranges up to 2000 microns does not exist; and

- Even if such instrumentation became availab le in the future, there would be no way to safely deploy it in the resulting dispersed oil plume by either ROV or research vessel without interfering with other critical, oil spill response operations.

With respect to droplet size determination, the Agenc y should focus on the most important aspect of subsea dispersant injection, which is to significantly reduce the initial droplet size to slow and change the surface expression of oil release as well as "thin out" the surface slick for better dispersion on the surface. Work by API and sponsored by BSEE suggest that HD high speed photography or sonar are the most promising methods of studying changes in droplet size during a full scale applic ation of subsea dispersants. We propose that alternative methods (i. e., high definition, high speed photography; sonar; etc.), as they mature and become available, should be allowed. Additionally, there should be flexibility to use other, improved scientific methods as they become available.

C. *Fluorometry is only useful as a screening tool.*

Indirect measures, such as fluorometry, are useful only as a screening tool to help refine monitoring locations and not as direct evidence of oil or specific oil concentrations in the subsurface as suggested.

In 300.913(b)(2), in addition to *in-situ* fluorometry, the Proposed Rule also requires the Responsible Party to conduct a fluorescence intensity analysis on collected water samples to determine fluorescence signatures of the dispersed oil for use as a relatively simple and rapid mea ns to assess dispersion effectiveness. Fluorescence intensity in this context is redundant and not as meaningful a measure as particle size analysis. An alternative would be to simply suggest this as a potential measure that could be used as the situation demands. Moreover, dispersant effectiveness assessments using SMART Tier 1 visual monitoring is a relatively simple, rapid way to evaluate this and is a well-accepted practice, as is the use of simple fluorometry from the SMART Tier II protocol for surface dispersant application. The Agency should substantiate the need to replace the tiered monitoring approach of the SMART protocols and, rather than listing all possible tests that can be conducted, specify how they add value to response decision-making and offer improvements over other, existing approaches.

D. *Detailed chemical analysis is unnecessary and many of the tests are too di fficult to perform in the field.*

In 300.913(b)(4), the Agency should recognize that more real-time operational monitoring information can be obtained by "quick screening" some samples for total petroleum hydrocarbons (TPH) by means of a hand-held gas chromatograph flame ionization detector (GC -FID). Detailed chemical analysis will not inform the response decision-making process and should instead fall under the purview of NRDA.

The requirement in 300.913(b)(5) for carbon dioxide measurements is unclear. While pH and dissolved oxygen monitoring seem logical, carbon dioxide (CO_2) measurements are not routinely made in deep water. It is not clear from the Proposed Rule what benefit carbon dioxide monitoring would bring that dissolved oxygen does not already provide. Similarly, the proposed requirement to measure methane is also unnecessary as it is also linked to potential oxygen deple tion, and the measure of DO is a reliable and sufficient method. There is a limited amount of sensors and water sampling devices that can be deployed from a research vessel at one time and inclusion of CO_2 detection complicates other elements for the deployment of the instrument array.

Regarding the DO measurements, we suggest simply requiring the *in situ* measurement of DO using best available technology (which would currently be automated sensors) to measure dissolved oxygen at depth and not expose it to surface pressures and temperatures before or during analysis. Verification can be achieved through the use of consistent sensor cleaning procedures, calibration tests, and redundant sensors which can be compared. Winkler titration, as discussed in the Proposed Rule, is an outdated

analytical method that is impractical for use on unstable offshore vessels, is prone to the introduction of human errors, and would slow the information flow within the response. If periodic bench tests are required to verify the integrity of the *in situ* DO sensor measurements, there are more consistent and reliable techniques than Winkler titration. We also recommend that only a fraction of the water samples be used to verify the *in situ* DO measurements as opposed to every sample , which will slow the process and information flow.

With respect to TPH analysis, the recommendation would be to provide these data for a fraction of the water column samples as opposed to every water sample (as currently proposed).

The requirement in 300.913(b)(7) for analyzing heavy metals has no bearing on dispersed oil monitoring. There is no operational reason for collecting these data. This should fall under the purview of NRDA. The source samples of oil are much better sources for determining the proportion of individual constituents. The carbon dioxide, methane, and heavy metals analysis requirements do little to inform response decisions, add unnecessary time and cost to the response activities, and would have a high degree of uncertainty in the absence of understanding the natural variations that might occur in the water column or in the case that a compounding discharge event (e.g., major flood) occurs during the response.

The focus of the Proposed Rule is almost exclusively on empirical data a nd should include some elements of 3-D and 4-D modelling to broaden the overall understanding of subsurface conditions.

We suggest revising the proposed language as follows:

> *(b) In areas not affected by the discharge of oil, collect a representative set of background water column samples following standard operating and quality assurance procedures, at the closest safe distance from the discharge as determined by the OSC and in any direction of likely transport considering surface and subsurface currents and oil properties for the variables listed below. Adherence to API TR1152 meets the requirements of this subsection. In the dispersed oil plume, at an interval designated by the OSC, collect water column samples following standard operating and quality assurance procedures, at such depths and locations where dispersed oil is likely to be present to analyze for the following:*
>
> > *(1) In-situ oil droplet size distribution, including mass or volume mean diameter for droplet sizes, with the majority of data coll ected between the 2.5 and 100 µm size;*
> > *(2) In -situ fluorometry signatures targeted to the type of oil discharged and referenced against the source oil;*
> > *(3) Dissolved oxygen (DO);*
> > *(4) Total petroleum hydrocarbons (based on an extracted water sample), individual resolvable constituents including volatile organic compounds, aliphatic hydrocarbons, monocyclic, polycyclic, and other aromatic hydrocarbons including alkylated homologs, and hopane and sterane biomarker compounds;*
> > *(5) 3-D and/or 4-D modeling when/where practical*
> > *(6) Turbidity;*
> > *(7) Water temperature;*
> > *(8) pH; and*
> > *(9) Conductivity.*
>
> *The purpose of analyzing these components is to map the extent and location of subsurface oil ; to characterize the condition of that oil ; and to help inform transport models that are also a key part of the decision-making process by helping to characterize the overall movement of subsurface oil.*

XVIII. Section § 300.913(c) Dispersant Effectiveness and Oil Distribution Characterization

A. *The Proposed Rule needs to be clearer on what is required for surface monitoring and what is required for subsea monitoring.*

Section § 300.913 provides monitoring requirements collectively for "subsurface use of dispersant in response to an oil discharge, surface use of dispersant in response to oil discharges of more than 100,000 U.S. gallons occurring within 24 hours, and surface use of dispersant for more than 96 hours in response to an oil discharge…" However, the preamble is often unclear as to whether every requirement within this section applies to surface or subsea dispersant applications or both. To facilitate a better evaluation of the proposed changes, each subsection should be divided into those aspects that apply to subsea applications and those to surface applications and those that apply to both. Elements within this section are already addressed in other sections of this Proposed Rule, particularly water column monitoring. The sections ought to be combined. The concept of an informed monitoring program (one that includes fixed and adaptive sampling locations and modeling) is already well understood and acknowledged in the spill response community.

In 300.913(c), requiring best available technologies may result in equipment not suitable for field conditions. Consideration should be given to changing "*available*" to "*practicable*."

In 300.913(c), the best available technology requirement should acknowledge aerial photography as a tool to measure effectiveness. This was a key method of effectiveness assessment during DWH and is not mentioned in the Proposed Rule. It is much more practical than the concepts put forward in the present proposal.

The relative effectiveness of the surface application should be determined by using the SMART protocols. For surface application, it is impossible to determine a numerical effectiveness as the amount of oil on the surface that is being sprayed is not known. Note that the analysis equipment often cannot return to the spray site in time to capture the information requested as the dispersant plume quickly dilutes or cannot be found.

XIX. Section § 300.913(d) Ecological Receptor Characterization

A. *Monitoring required to determine possible environmental effects is too time consuming to support dispersant operations decisions.*

Industry agrees that environmental monitoring should be performed to support operational decision-making. Monitoring required to determine environmental effects, however, is likely to be too time consuming to support dispersant operations decisions and should remain a NRDA activity.

This section of the Proposed Rule requires the Responsible Party to characterize the ecological receptors and to consult with the OSC to determine an acute toxicity level of concern for the dispersed oil using available dose/response information relevant to potentially exposed species. Conducting the required ecological characterization of the spill site may not be possible in the available response time-frame. If the untreated oil is likely to drift ashore and impact a sensitive coastal resource within a day or two unless it is dispersed, there will be a very finite period of time for such considerations suggested in the Proposed Rule.

With respect to the proposed ecological thresholds, although it is constructive to have known benchmarks, it is not clear what decision would be forthcoming if a threshold were exceeded and would certainly be misplaced to derive new thresholds during a response. The documented toxicity of dispersed oil to individual laboratory organisms is by no means a measure of the overall population of that

organism, nor of the ecosystem as a whole, to recover from oil exposure. During a spill, dispersants operations might be stopped because of the threshold of an individual species even though there may be the potential for higher environmental damages beyond that species. During a spill, dispersants operations might be stopped based on a threshold for an individual species . The suggestion of using species sensitivity distributions (SSD) during an active response is a misuse of the method (which requires long periods of time to assess) and is counter to establishing frameworks that are relevant in the dynamic ocean setting.

Rather than requiring the Responsible Party to make this real-time assessment, a CERA/NEBA process (conducted as part of contingency planning) should serve as the basis to make operational decisions on whether dispersants and/or other agents should be used during a response. A CERA/NEBA is the only mechanism that allows a full environmental trade-offs analysis of all response options to be conducted.

XX. Section § 300.913(e) Immediate Reporting Requirements

A. *Reporting requirements should be more flexible.*

The Proposed Rule does not include any indication of what happens if such a requirement to immediately report the information is not met for any reason. From a functional standpoint, the idea of reporting is sound, but the OSC should have the discretion to determine the frequency of reporting.

The Agency is also requesting comments on whether similar requirements should be applied to surface dispersant application. There are already practical and agreed reporting requirements for surface dispersant use and the natural weathering and dispersion that takes place at the surface is fundamentally different than what might occur in the subsurface.

XXI. Section § 300.913(e)(1) Daily Sampling and Data Analyses Reporting

A. *The Agency should not develop requirements for daily authorizations of dispersant quantities.*

This section requires reporting of a deviation of more than 10 percent from the mean hourly volume of dispersant authorized for "*24 hours use*." The Proposed Rule does not require authorization of dispersant use in 24 hour increments. Such restrictive authorization requirements tremendously complicate dispersant operations while providing little environmental benefit. The Agency should not develop requirements for daily authorizations of dispersant quantities, as this decision should be based on the ongoing spill scenario and should be left to the OSC. Dispersant application rates may need to vary to account for dynamic spill conditions, environmental conditions, and other ongoing response operations. Placing daily limits on volumes of dispersants that may be used prevents trained response personnel from utilizing appropriate dispersant application rates. Furthermore, it circumvents the NEBA process that evaluates the best response options that protect the environment as a whole.

We suggest modifying the proposed language as follows:

> *(e) Report to the OSC as soon as practicable or at an interval determined by the OSC:*
> *(1) Deviation of more than 10 percent from the mean hourly dispersant use rate for subsurface application, based on the dispersant volume authorized for 24 hours use, and the reason for the deviation; and*
> *(2) Any threatened or endangered species, or other priority ecological receptors formally identified by the OSC in consultation with the Environmental Unit, that may be exposed, based on dispersed plume trajectory modeling and level of concern information.*

XXII. Section § 300.913(e)(2) Immediate Reporting of Ecological Receptors

A. *Real-time ecological receptor analysis is unrealistic and should be part of a CERA/NEBA process.*

The requirement in this section for real -time ecological receptor analysis is unrealistic. Such efforts need time and broad stakeholder participation to be meaningful. This information should be part of a NEBA process to document the resources at risk of the potentially affected ecosystem. Only through this detailed NEBA process can an adequate understating of overall ecosystem recovery be developed and assessed. Further, the process of overall environmental trade -offs is independent of individual species impacts and should be based at the ecosystem level.

XXIII. Section § 300.913(f) Daily Sampling and Data Analyses Reporting

A. *Daily reporting of water sampling data should only apply to subsea dispersant injection and is not useful for dispersant decision-making.*

This section should be amended to clarify that daily reporting of water sampling data only applies to subsea dispersant injection. Further, the requirement in Section 300.913(f)(3) for a 5-day turnaround time for analysis is unrealistic, given that it could take several days for the sample to be transported to shore and additional time for it to reach a qua lified laboratory and put in the queue for analysis. During DWH, the sheer quantity of samples overwhelmed both the regional and the nation's labs processing abilities. In many cases, the backlog of samples lasted 30 days or longer. A blanket requirement o f five days is unrealistic and should be replaced with a negotiated time that is staggered on the basis of sample data priority for operational decision-making. The data may be useful for NRDA, but not for dispersants decision-making because other monitori ng techniques can better inform decision-making. These water samples will provide information about concentrations in water from 5 or more days previously, but will be irrelevant for operational questions that require near real-time feedback.

We recommend combining this section with the proposed "*Immediate Reporting*" section into one section that notes the importance of reporting in supporting the coordination of response actions. Suggest modifying proposed language as follows:

(e) Reporting

(1) Report to the OSC as soon as practicable or at an interval determined by the OSC any:

(i) Deviation of more than 10 percent from the mean hourly dispersant use rate for subsurface application, based on the dispersant volume authorized for 24 hours use, and the reason for the deviation; and

(ii) Ecological receptors, including any threatened or endangered species that may be exposed based on dispersed plume trajectory modeling and level of concern information.

(2) Report to the OSC at an interval determined by the OSC, water sampling and data analyses collected in § 300.913(b) and include:

(i) Specific hourly dispersant application rate and the total amount of dispersant used for the previous reporting period established by the OSC with concurrence from the EPA representative to the RRT;

(ii) All collected data and analyses of those data within a timeframe necessary to make operational decisions, including documented observations, photographs, video, and any other information related to

dispersant use, based on a reasonable and appropriate timeframe authorized by the OSC;
(iii) Estimates of the transport of dispersed and non-dispersed oil and associated volatile petroleum hydrocarbons, and dispersants, using the best available trajectory modeling.

XXIV. Section § 300.915(a) General Information for any Product Category

A. *The proposed information requirements to be submitted by dispersant manufacturers may reduce the number of dispersant suppliers.*

The proposed requirements for information to be submitted, most likely by dispersant manufacturers, as described in 300.915 (a)(1)-(21), are extensive. API and NOIA are concerned that dispersant manufacturers may not have a sufficient business case to meet all proposed data requirements. The Agency therefore needs to demonstrate that this Proposed Rule will not collapse the market for dispersants and drive away all viable producers from supplying products to meet the need. Further, the Agency should consider developing a total re-listing cost analysis.

XXV. Section § 300.915(a)(5) Safety Data Sheet

A. *A Safety Data Sheet for each component in a product is unnecessary.*

The proposed requirement includes *"A Safety Data Sheet ('SDS') for the product."* We believe this is a reasonable request. However, in the preamble, the Agency solicited comments on the value of requiring an SDS for each individual component. We believe that requiring more than just an SDS for the whole product will add unnecessary burden to the submitter that will not add any protection. The information needed to protect responders will be provided in the SDS for the product as a whole.

XXVI. Section § 300.915(a)(11) Environmental Fate Information

A. *Vendors should not be required to generate new environmental fate information for their products.*

The proposed language in this section should be modified to clearly state that this data should be reported *only* if it is already available. As currently written it is unclear if the vendors are being asked to collect new data for their products to cover these parameters.

XXVII. Section § 300.915(a)(13) Identity and Concentration of Product Components

A. *Vendors should not be required to provide the identity and concentration of product components.*

Some vendors may refuse to submit this detailed data as it may threaten a proprietary advantage. This may result in less favorable/effective product being listed in the Product Schedule. We recommend that reporting should be restricted to naming the ingredients, but should not require that the percentage of each ingredient is disclosed. Although Proposed Rule Section 300.950(b) allows this information to be claimed as confidential business information, the requirement under 300.915(a)(13) still presents challenges because there is no guarantee that the claim of confidentiality will be accepted. This may discourage manufacturers from supplying dispersant products.

XXVIII. Section § 300.915(a)(15) Certification

A. *The requirement that product not contain levels exceeding the National Water Quality Standards is unnecessary.*

The agency is proposing certification, including data, methodology, and supporting documentation, indicating that the product does not contain levels that exceed the National Water Quality Standards lowest aquatic life acute value for the following contaminants: metals reasonably expected to be in the product including arsenic, cadmium, chromium, copper, lead, mercury, nickel, vanadium, zinc; cyanide; chlorinated hydrocarbons; pesticides; polychlorinated biphenyls (PCBs); and polynuclear aromatic hydrocarbons (PAHs).

The National Water Quality Standard Contaminants are defined as concentrations in the water column. Water quality criteria are applicable to receiving waters, not formulated products, and this proposal assumes the appropriate exposure is to full-strength (100%) product rather than a reasonable environmental-realistic exposure concentration.

This requirement is not reasonable because of the enormous dilution that occurs when any product is used in an oil spill situation, and it is not required for other chemicals released into the environment. We believe that the existing requirements to communicate hazardous impurities on product SDSs should be sufficient and suggests removing the requirement.

XXIX. Section § 300.915(a)(19) Annual Product Production Volume

A. *The requirement for vendor production capability is not realistic of the market for dispersants.*

Most dispersants are blends of components and are only manufactured upon request. Capacity is not fixed and varies with available blending tankage, existing business demands, other product orders, and component supplies/shipping constraints. While such data are very useful in a large spill where existing stockpiles could be depleted, the information provided in such a request at the time of the application would not be relevant to a future time when product manufacturing could be required during a response. Further, the development of both U.S. stockpiles and international Dispersant Global Stockpiles were designed to ensure that sufficient, effective dispersants will be available when needed. We believe this requirement should be removed. If not removed altogether, the Agency should consider modifying the language to require dispersant or agent producers to be able to provide such information within 24 hours should the OSC need it and make it a condition of product authorization. If the Agency is concerned about the availability of sufficient volume of dispersants required for operations, it should ensure that the main dispersants products that are used around the world and are extensively studied are included on the National Product Schedule and are available for response in the U.S.

XXX. Section § 300.915(a)(20) Recognition

A. *Design for Environment recognition is unnecessary.*

The Agency asks for recognition from EPA's Design for the Environment (DfE) if applicable. DfE recognition should not be included in the Proposed Rule because it is not required, and the Agency has other ways to advance voluntary programs.

XXXI. Section § 300.915(b)(1) Dispersant Efficacy

A. *The new efficacy test cannot predict field dispersant effectiveness.*

API and NOIA support the Agency's proposed change in efficacy testing from the Swirling Flask Test (SFT) to the Baffled Flask Test (BFT) method. The SFT was a very low energy test and required the use of low viscosity test oil s to achieve significant levels of measured dispersant effectiveness. The BFT is a much more energetic test method thus permitting the use of higher viscosity test oils to achieve higher levels of measured effectiveness with the same dispersant. It is impo rtant to recognize that such laboratory tests can evaluate relative effectiveness between different dispersants, but cannot be used to predict the effectiveness of dispersants in the field on specific oils under specific conditions , such as differences in wave height, possible application in high turbulence subsea or using prop wash in ice - covered waters.

B. *The new test oils may not provide comparable results if new test samples are obtained.*

We also support the Agency's proposed changes in test oils from South Louisiana Crude (SLC) and Prudhoe Bay Crude (PBC) to Alaska North Slope (ANS) and Intermediate Fuel Oil (IFO–120), since these new test oils provide a wider range of properties in the oils to be tested. The agency should note that ANS is a blend of c rude oils produced on the North Slope of Alaska, and the composition and properties have varied through the years as the crude oils produced on the Alaska North Slope have changed . Also, IFO-120 composition is entirely dependent on the crude run at the supplying refinery and their blending needs. However, IFO-120 is a blend of residual fuel components including vacuum resid uum cut with a lighter product to achieve a specified viscosity (i.e. , 120 cSt at 100F in this case), but will vary dramatically in composition between lots at a given refinery and between refineries. It is probable that the varying composition of IFO-120 will result in inconsistent DE results over time. This means that a single batch of IFO-120 will have to be preserved for all future pro duct testing to ensure comparability. The Agency should conduct stability tests of ANS and IFO -120 stocks to ensure they can maintain viable samples over time. The Agency could consider substituting a heavy crude oil rather than this heavy refined product or store a large single batch of IFO -120 with data provided on shelf stability or a provision to account for changes over time.

C. *Testing dispersant efficacy at different temperatures will not provide information that will predict dispersant effectiveness in real world conditions.*

We oppose introducing testing at different temperatures. This new approach implies that laboratory tests represent field conditions in some way which is not the case. The Agency proposes that dispersants should only be approved for the type of oil they worked in the lab and temperatures tested in the lab. This is incorrect as:

- different oils will be treated than the two tested in the laboratory;
- even the same oil will weather with potential effects to dispersant efficacy;
- turbulence in environment will surely be different;
- subsea injection even into the same oil in the Arctic is different because of oil temperature and turbulence level subsea; and
- dispersing oil in ice is also different from lab tests because of the mixing provided by vessel propellers.

In the preamble to this same section, the Agency appears to suggest that the Proposed Rule changes are intended to indicate the likely effectiveness of operational dispersant use on oil spilled in the Arctic or Gulf of Mexico (column 3, 80 Fed. Reg. 3403; both locales are specified). Relatively simple efficacy tests such as the BFT are incapable of simulating the range of parameters and processes that will be encountered and influence real -world dispersant effectiveness. The Rule sho uld clearly state that laboratory efficacy testing is a comparative screening tool for different dispersants under laboratory conditions and should recognize that a ctual dispersant effectiveness on oil treated at sea can vary a great deal from

effectiveness data collected in a laboratory. A dispersant/oil combination that produces an effectiveness result in the BFT laboratory method of 70% might result in >95% dispersion at sea (e.g., in a moderate sea state), and less dispersion in other circumstances (e.g., calmer sea state). Therefore as part of dispersant use authorization, divorced from the product approval process, the availability of suitable mixing energy becomes an operational consideration, coupled with the dilution potential for the dispersed oil. We suggest that reference to specific regions should be deleted from this section.

D. *Dispersant efficacy thresholds listed in the Proposed Rule are too high.*

In the preamble to this section (column 2, 80 Fed. Reg. 3403), the Agency's justification for dispersant efficacy thresholds (DE values) includes:

> "*Three of the eight dispersants tested clearly differentiated themselves from the other five dispersants for having the best DE_{LCL95} efficacy results… This natural break in the results provided the Agency with the basis for the proposed threshold criteria.*"

We believe the proposed thresholds of efficacy for listing on the Schedule are high. The supporting documentation in the docket shows that only six (not eight) dispersants were tested and two of these six dispersants were only tested with one oil. The rationale for selecting the proposed efficacy thresholds for listing on the Schedule is not convincing on the basis of the limited data presented. Data from another EPA study[31] shows significant variability in the results obtained with the BFT. Variability in test results is troubling and this variation could lead to confusing data sets where the same dispersant sometimes "passes" and sometimes "fails" because of the high threshold limit that has been set. While the Agency states it tested the dispersants currently listed and found three of them to be sufficiently effective, the EPA study on the BFT funded by BSEE and published in 2011 suggests otherwise. As shown in Table 3 of this BFT study, Corexit 9500 may not be listed based on published Dispersant Effectiveness (expressed as the LCL95) of 69% and 61% on ANS Fresh and IFO-120, respectively, at room temperature (15° C). This suggests that the recommended thresholds are potentially too restrictive and do not sufficiently take into account the variability of the BFT.

The Agency should reconsider the proposed efficacy thresholds for listing on the Schedule. We suggest that these thresholds could be lowered without any significant impairment to the effectiveness of operational dispersant use at a response. In addition, reporting only the DELCL95 reduces the amount of information available on a product. We recommend that the test average and standard deviation should be provided, which would provide additional information on the precision of the testing.

The Agency makes the following statement in the preamble (column 1, 80 Fed. Reg. 3404):

> "*As a result of this increased mixing energy, better dispersion is realized under conditions more realistic of wave action in the sea*"

We believe that this statement would be more accurate if it were re-phrased as follows:

> "*As a result of this increased mixing energy, higher dispersant effectiveness results are obtained.*"

Since the recent subsea dispersant injection development does not have major implications for the rationale behind the current tests for dispersant product approval, it might be important to recognize that the laboratory test data are sufficient to identify whether a product has appropriate effectiveness,

[31] Venosa and Holder, 2011, Laboratory-Scale Testing of Dispersant Effectiveness of 20 Oils Using the Baffled Flask Test

irrespective of whether the product may be considered for use in surface applica tion or subsea injection (SSDI) scenarios. Additional testing for SSDI approval is not necessary.

XXXII. Section § 300.915(b)(2) Dispersant Toxicity tests and listing criteria

We offer the following comments on the proposed revisions to the toxicity testing and has ordered the comments by specific test.

A. *Standard static acute toxicity testing should be used only for the dispersant alone.*

The Agency proposes that a dispersant shou ld be tested using a modified standard static acute toxicity test for the dispersant with *Americamysis bahia* and *Menidia beryllina* using the 48 hour and 96 hour duration LC_{50} procedure. We agree there is a need for toxicity testing of dispersants to preven t highly toxic (LC50 <1ppm) dispersants from being listed on the Schedule. The proposed toxicity test method, although designed for effluent testing and having no relevance to real -world exposures of marine organisms to dispersants during dispersant use on oil releases, is a convenient way to screen individual dispersants. We agree with this proposal for dispersant-only toxicity tests.

The Agency also specifies a threshold for a dispersant to be listed on the Schedule, *"the LC_{50} being that at the lower 95% confidence interval for all acute toxicity tests must be greater than 10 ppm."* We believe this is a reasonable threshold for the "dispersant only" toxicity test.

B. *Standard static acute toxicity testing of dispersant/oil mixtures does not represent real world exposures.*

The purpose of the proposed toxicity testing of dispersant/oil mixtures is not clear. If the objective of the toxicity testing is to ensure that any dispersant that is to be listed has less than a specified level of toxicity measured in the test, then a dispersant alone toxicity test is sufficient.

We oppose the proposed dispersant/oil acute toxicity testing as a requirement for a dispersant to be listed on the Schedule, for the following reasons:

- It is now widely accepted by scientists that any concerns about the potential for toxic effects on marine orga nisms resulting from the use of modern dispersants should consider the potential effects of dispersed oil, not the dispersant itself. This is logical because dispersant is used at low concentrations, compared to the dispersed oil concentrations produced by their use , and they have much lower toxicity than oil . The potential severity of toxic effects that could be caused to marine organisms by exposure to dispersed oil will be a function of exposure: the concentration of dispersed oil (or oil components) in the water, the duration of that exposure , and oil type and receiving organisms . The Agency proposed dispersant/oil mixture tests will not provide any information about real-world exposure.
- More sophisticated dispersant/oil mixture toxicity testing procedu res, using more relevant exposure regimes, can produce information about the potential for toxic effects on marine organisms by oil dispersed (or oil compounds released in to the water) as the result of dispersant use. Such toxicity studies may be relevant to deciding where and when dispersant use may be authorized, but not to dispersants being listed on the Schedule.
- Toxicity testing of dispersant/oil mixtures (i.e., dispersed oil) usi ng the proposed standard static acute LC_{50} procedures will inevitably discriminate against more effective dispersants which disperse more oil into the water, thereby increasing exposure to aquatic organisms. Such testing does not give any indication of the potential for toxic effects to be caused by dispersant use on spilled oil on the sea surface.

- Previous work on dispersant/oil mixture toxicity testing conducted by the Agency in 2010 (Comparative Toxicity of Louisiana Sweet Crude Oil [LSC] and Chemically Dispersed LSC to Two Gulf of Mexico Aquatic Test Species [published by Hemmer, Barron and Greene]) shows that the proposed 10 ppm threshold (determined with LSC oil, not ANS or IFO -120) with *Americamysis bahia* would not be met by any of the eight dispersa nts tested at that time. Only one dispersant would have met the 10 ppm threshold with *Menidia beryllina*.

In general, the Agency should clearly distinguish between the requirements of (i) the toxicity testing required to assess which dispersants should be listed on the Schedule, and (ii) toxicological studies with appropriate oils, test organisms and exposure conditions that will inform discussions about how the listed dispersants might cause impacts in U.S. waters under the specific circumstances of an oil spill or release. The two toxicity test approaches required are incompatible. Attempting to use the same standard, static acute LC_{50} procedures to provide information or guidance on both aspects will not achieve these goals. Since the proposed dispersant/oil mixture toxicity testing requirements in this section do not address item (ii) above, the Agency needs to clarify the objective and rationale of the proposed acute exposure toxicity testing of dispersant/oil mixtures and should explain how this relate s to the listing of a product on the Schedule.

C. *Testing of dispersants using the developmental assay for the purple sea urchin is not relevant.*

The Agency proposes a requirement for an EPA test protocol involving sea urchin embryos, routinely used in eff luent testing, to be used to assess the potential for a dispersant product to cause adverse effects on the developmental process. The relevance of this test, which was developed for the assessment of wastewater effluents, to dispersant use is highly questi onable. Again, trying to estimate potential real world consequences to dispersed oil exposure should be part of a NEBA decision -based process. The environmental trade-offs of using dispersants in offshore environments to protect sensitive shoreline habitats from oil exposure is being circumvented. Given that the purple sea urchin is found in shallow coastal waters, there is no relevance of applying purple sea urchin data given that dispersant operations are unlikely to occur in these environments. In additi on, the proposed threshold that the LC_{50} must be greater than 10 ppm is not a relevant indicator to inform any potential real -world effects of dispersant use in the environment. Additionally, if the agency applies their proposed toxicity thresholds to the sub-chronic sea urchin toxicity tests, it is likely that many (maybe all) current and approved dispersants in the U .S. will fail the threshold . Some of them could also fail with the sub -chronic *M. beryllina* tests.

We oppose the proposed developmental ass ay and do not believe it should be a requirement for a dispersant to be listed on the Schedule. API and NOIA are also concerned that little experience is available in application of this assay to dispersants, and it is unclear about the intra -lab precision associated with this proposed requirement and the practical challenges that may arise in interpretation of test results.

D. *Sub-chronic 7 -day static toxicity tests for dispersant are not relevant and are unnecessary.*

The Agency states (300.915(b) (2), 80 Fe d. Reg. 3405) that it " *has limited information concerning the possible sub -lethal effects of dispersants currently listed on the Product Schedule* " and that " *This information would also be of value as guidance to regional responders and OSC's on possible ad verse effects on survival and growth of larval fish and invertebrates caused by longer -term exposure to dispersants.*"

Chronic exposure to dispersant alone is not a relevant indicator of risks posed when dispersant is used to mitigate the effects of an oil release. As stated previously, this test was developed and validated for wastewater effluents and is not relevant to the use of dispersants in offshore spill response. Rather than proposing sub-chronic toxicity testing requirements for any dispersant to be listed on the Schedule, the Agency could undertake studies to establish whether this is a valid concern, or whether it could provide useful information under the likely conditions of dispersant use on offshore oil releases.

The Agency's proposal of a No Observed Effect Concentration (NOEC) equal to or greater than 1 ppm listing threshold level for sub-chronic effects is arbitrary. Although this threshold level provides a ten-fold safety factor over the proposed 10 ppm acute toxicity threshold values, it could also be questioned on the same basis; static toxicity tests over 48 to 96 hours do not represent relevant exposure scenarios in the marine ecosystem following offshore applications of dispersant.

We oppose the proposed sub-chronic toxicity testing and does not believe it should be a requirement for a dispersant to be listed on the Schedule. Sub-lethal/sub-chronic studies are not necessary (acute toxicity endpoints are considered relevant) since the global commercial dispersant products are confirmed as biodegradable so that the surfactants and solvent constituents of the dispersant formulation do not persist in the marine environment causing adverse effects. The French dispersant product approval regulator requires a biodegradability test of the dispersant as part of its product approval process. Commercial dispersants commonly stockpiled around the world are therefore readily biodegradable and pass the French test. If a dispersant is approved in France, it must have passed the biodegradation test (AFNOR standard NF T90-346). It is recommended that ready biodegradability of a dispersant be required for approval in the U.S. as well. Therefore any candidate dispersant product would need to be tested for biodegradability, or the dispersant manufacturer could be required to state that the formulation constituents are biodegradable and provide supporting evidence.

E. *Dispersant testing should only be used for screening individual dispersant products.*

If the objective of the toxicity tests are to select less toxic dispersants (based on an aquatic toxicity threshold), then dispersant alone toxicity tests are sufficient and the maximum toxicity threshold of a candidate dispersant should be set if the dispersant is significantly less toxic than the reference oils. The dispersed oil toxicity testing (CEWAF) should not be considered for dispersant use authorization as it discriminates against the effective dispersant products which disperse more oil into the water thereby increasing exposure to the aquatic test organisms. It is also important to note that CEWAF toxicity testing does not simulate actual dispersant use on spilled oil in the marine environment. Additionally, static toxicity tests over 48 to 96 hours do not represent relevant exposure scenarios in the environment following surface application of dispersants.

In summary, the Agency's suggested toxicity testing changes in the Proposed Rule do not take into account the essential role of NEBA in considering the potential benefits and risks of dispersant use as a response option for an offshore oil release. The Product Schedule should focus on the basic testing to compare the efficacy and toxicity of the dispersants against each other with the sole purpose of screening good products that should be listed. The Proposed Rule should not attempt to translate any laboratory data into possible real-world implications. This process should instead be defined by full stakeholder NEBA discussions specific to a given spill scenario.

XXXIII. Section 300.915(c)(1) Surface Washing Agent Efficacy

A. *Surface Washing Agent (SWA) efficacy testing should not be required until the Agency develops its own test protocol.*

In the preamble the Agency has requested comments on available methodologies and whether the rule should identify the specific methodologies to be used until the Agency develops and adopts a new testing protocol for SWA efficacy testing (column 1, 80 Fed. Reg. 3407). The Agency lists two tests by ASTM and Environment Canada as possible interim tests while the Agency develops its own test. There is no evaluation of how the ASTM test compares to the Environment Canada test. As a result, expecting similar results by both tests is unrealistic. The average efficacy in the published data reported using the Environment Canada test may indeed be approximately 30%, but the variation is high and the tests were performed on a Canadian oil using only one test. We believe the Agency should delay requiring SWA efficacy testing until it develops its own test protocol that utilizes a U.S. oil.

XXXIV. Section § 300.915(d) Bioremediation Agent Testing and Listing Requirements

A. *An additional protocol specific to products containing enzymes only would be appropriate.*

The Agency is requesting comments on whether an additional protocol specific to products containing enzymes only would be appropriate. This would consist only of exposure water, weathered oil and the enzymatic product in the concentrations specified by the manufacturer. A new protocol would be useful because effectiveness data from the testing would help determine whether this technology would be beneficial during a response. Many claims may be made about "safer" products to treat oil and thus more credible scientific information is warranted on products approved on the Schedule.

XXXV. Section § 300.915(e) Solidifier Testing and Listing Requirements

A. *Solidifier toxicity testing would be useful.*

The Agency is requesting comments on the need for acute toxicity tests conducted with solidifiers - reference oil mixtures. Toxicity tests with oil may be beneficial as they will help to evaluate efficiency of solidifiers in retaining water soluble hydrocarbons and preventing them from leaching into water. Simple efficiency tests may not provide such data.

XXXVI. Section § 300.915(g) Sorbent Requirements

A. *The proposed Sorbent Product List should not exclude natural sorbent materials.*

The Agency proposes to make available a Sorbent Product List that includes certain publically available non-proprietary sorbent materials. We believe this should be a generic list and should not exclude natural sorbent materials (e.g., hay) that could be obtained locally at a spill site.

XXXVII. Section § 300.950 Submission of Confidential Business Information (CBI)

A. *Full public disclosure of product information may reduce the number of products being made available in the U.S.*

In 300.950 (column 3, 80 Fed. Reg. 3426), the Agency requires everything in the product listing be made public except the actual concentrations of product ingredients. While the Agency may need full disclosure of product formulae, we do not believe the Agency needs to make them public and can accept products with both the ingredients and the concentrations as CBI. Any disclosure of information on product formulae (either ingredients or % concentrations) could put a manufacturer at a competitive disadvantage. This may discourage some manufacturers from listing their product at all and could result in effective dispersant products not being available in the United States.

XXXVIII. Section § 300.955 Addition of a Product to the Schedule

A. *The transition period for currently listed products is too short.*

In 300.955(f), the Agency specifies a 24 -month transition period for currently listed products. This is a very short timeframe given the variety of products listed and depth of technical work (e.g., toxicology studies) required to be contracted, conduc ted, analyzed, and reported. We believe the Agency should extend the transition period to the lesser of five years, the current product expiration date, or after suitable replacement products are listed and commercially available.

We also believe that this section should include a discussion of how currently owned stockpiles purchased in 2010 and 2011 (more than one million gallons of Corexit, in the U .S.) will be tested to comply with the proposed existing stockpile testing requirements of being tested every five years. It is recommended the Agency state that currently held stockpiles will have five years to be re-tested after that dispersant product is listed on the new Product Schedule.

XXXIX. Appendix C to Part 300 Requirements for Product Testing Protocols and Summary Test Data

A. *The dispersant-to-oil ratio (DOR) ratio used in the BFT should be 1:20.*

In Appendix C, Section 2.1 (column 3, 80 Fed. Reg. 3428), the BFT mandates a DOR of 1:25. A DOR of 1:20 has commonly been accepted for use in the U.S. and abroad. Further, the 1:25 DOR may not even be within the manufacturer's recommended use ratio. The Agency needs to ju stify this test requirement. We believe that the DOR for the BFT should be changed to 1:20 to align with the regulatory requirements in 33 CFR 154 and 155 for calculating aerial and vessel dispersant capabilities and to align with standard dispersant spray system calibration. A DOR of 1:20 leads to a 5 gallon per acre dosag e for surface spray systems and is the minimum amount of dispersant to oil slicks that can be visually identified as having sufficient thickness to warrant dispersant application. Having the testing procedure completed for the same DOR would provide more reliance that, when applied, the dispersant will be effective.

B. *The Agency should explain how it will address changes in the refe rence oils as they age and as new samples are acquired for the future.*

In Appendix C, Section 2.3.2, the Agency requires the u se of ANS and IFO -120 as standard reference oils. The Agency is supplying these materials to labs but no data is presented showing shelf stability over time for either and whether stable equivalent materials can be provided in the future. No provision is provided for addressing changes in dispersant efficacy due to changes in composition of ANS (e.g., field production changes, new fields in pipeline blend, etc.).

Also, IFO -120 composition is entirely dependent on the crude run at the supplying refinery an d their blending needs. It is a blend of residual fuel components including vacuum residue cut with a lighter product to achieve a specified viscosity (i.e., 120 cSt at 100 degrees F in this case), but it will vary dramatically in composition between lots at a given refinery and between refineries. If multiple IFO -120 samples were provided from different refineries to a toxicity laboratory, there would be varying results because IFO-120 composition changes from day to day and between refineries. This means that a single batch of IFO -120 will have to be preserved for all future product testing to ensure comparability. The Agency should conduct stability tests of ANS and IFO -120 stocks to ensure they can maintain viable samples over time.

CONCLUSIONS

In conclusion, as evidenced by the extensive substantive comments provided herein, it is clear that much work needs to be done before the Agency should finalize a rule in this key area of spill response, including to make sure that the final rule does not have the unintended consequence of eliminating all or most commercially available and future dispersants for use in spill response. We believe the Proposed Rule will likely result in dispersants being eliminated as a response technique, for the following reasons:

- The Proposed Rule does not recognize that response option decisions must be based on NEBA and that dispersing oil in offshore environments , where it may dilute and biodegrade, is often preferred over the long -term environmental damages that will occur when untreated oil strands and persists in shoreline environments;
- The Proposed Rule focuses on potential risks of dispersants without recognizing the primary purpose of dispersants, which is to rapidly dilute the spilled oil and enhance removal of the o il from the environment through microbial degradation, resulting in reduced VOCs (personnel safety), and a reduction in persistent, stranded oil in shoreline ecosystems (leading to faster ecosystem recovery);
- The Proposed Rule's dispersed oil toxicity tes ting thresholds could eliminate all dispersants that are on the current NCP list and could preclude the listing of future dispersants that are currently in development. Furthermore, the proposed dispersed oil toxicity testing thresholds favor ineffective dispersants over effective products and will mistakenly measure the toxicity of the standard test oils, not of the dispersant product;
- The Proposed Rule adversely constrains the ability of the OSC to make the necessary operational decisions to ensure for an expeditious, efficient, and effective response to mitigate the impacts of a spill to the greatest extent possible;
- The Proposed Rule raises significant issues related to USCG and BSEE facility and vessel oil spill response plan compliance and could resul t in unnecessary and increased environmental impacts and corresponding spill claims as a result of increased damages;
- The Proposed Rule's stockpile re-testing procedures have not taken into account manufacturer recommendations, nor considered scientific publications on the demonstrated, lengthy shelf-life of dispersant products; and
- The Proposed Rule's dispersant monitoring requirements are not focused on collecting data that can provide useful information regarding an ongoing dispersant operation. Instead, these requirements have the potential to delay dispersant use resulting in unnecessary environmental damage.

In addition, the RIA conducted by EPA is substantially flawed. The RIA falls woefully short of EPA's regulatory requirements. EPA fails to provide any quantification of the potential social benefits of the Rule; in fact, the RIA lists benefits associated with avoiding and limiting the impacts of oil spills as potential benefits of this Rule, even though the Proposed Rule has the potential to ag gravate the impacts of an oil spill by limiting response options. As a result, the RIA significantly underestimates the potential costs of the Proposed Rule , while also underestimating the direct costs that the Rule would impose on small businesses, government, and the greater industry.

Accordingly, API and NOIA strongly urge the Agency to consider these comments and suggested revisions to the Proposed Rule to ensure that the best dispersants and dispersant-use practices are available for offshore oil spill response. We recommend that there be further discussions between the Federal agencies and industry scientific and operational experts to add value to further revisions of the Proposed Rule to ensure that the proposed changes to Subpart J align with the latest scientific practices and do not artificially bias against the use of certain response options, such as dispersants.

Furthermore, API and NOIA recommend EPA initiate a Negotiated Rulemaking which has proven to be effective in controversial rulemakings such as this one. At a minimum, we urge the Agency to publish a Supplemental Proposed Rule in the Federal Register for additional public comments before publishing a final rule to ultimately make sure that responders, both from the government and industry, are able to use dispersants in the most effective and expeditious manner, taking into consideration necessary environmental and safety factors, to provide for the best response possible depending on the circumstances of a particular spill event. Given the time it will take to initiate and complete a new rulemaking after the final rule is published it is imperative for the general public and all stakeholders that we get it right the first time.

Thank you for your time and attention to these comments on the Proposed Rule. API, NOIA and our member companies share EPA's goal of establishing sound Dispersants Use Policy and are committed to working with the Agency and the other Federal Agencies involved in oil spill response to ensure the health and safety of people and minimize environmental and community impacts during an oil spill. Please do not hesitate to contact Emily Hague (202-682-8260, Hague@api.org) or Nicolette Nye (202-465-8463, nnye@noia.org) if you have any questions.

Sincerely,

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Attachment A

Analysis of EPA's Regulatory Impact Analysis for the Proposed Revisions to the National Oil and Hazardous Substances Pollution Contingency Plan in 40 CFR 300 Subpart J

Submitted by:

Policy Navigation Group

April 2015

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EXECUTIVE SUMMARY

Policy Navigation Group (PNG) was asked by the American Petroleum Institute to review the Regulatory Impact Analysis (RIA) for U.S. Environmental Protection Agency's (EPA) proposed rule entitled *National Oil and Hazardous Substances Pollution Contingency Plan.*[1] We were asked to evaluate whether the RIA for the proposed rule is consistent with the analytical standards set forward in OMB Circular A-4 and EPA's Guidelines for Preparing Economic Analyses and to perform an independent cost analysis.[2] In our analysis, the proposed RIA has the following principal deficiencies:

- The proposed rule's RIA does not have a consistent baseline that allows the public and policy officials to compare the incremental social costs and the incremental social benefits of EPA's proposed regulation and the individual policy options within the proposal.
- The RIA fails to estimate many policy scenarios and requirements issued in the proposed rulemaking. The RIA thus fails to provide the public a full description of the proposal's impact and undermines the public's opportunity to provide meaningful comment on the proposed rule.
- Further, for the provisions that it does give quantified social cost estimates, the RIA often underestimates compliance costs. The RIA thus underestimates the full social cost to comply and to administer the changes for both private firms and government agencies.
- The RIA does not provide any quantified social benefit estimates either for the proposal as a whole or for its individual options. Moreover, its qualitative description of social benefits is inaccurate and at times actually describes social costs of the proposal. Without estimates of the incremental social benefits of the rulemaking and the proposal's regulatory options, the public cannot provide meaningful comments on the merits of EPA's proposal.
- The RIA's methodology is designed to underestimate the small business impact. EPA's economic impact analysis is inconsistent with EPA's economic analysis guidelines and EPA's requirements under the Regulatory Flexibility Act (RFA).[3] In this analysis, we address some of the RIA's deficiencies and find that the rulemaking could have a significant impact on a substantial number of small businesses subject to the rulemaking.

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[1] 80 FR 3380. Also, U.S. Environmental Protection Agency, *Regulatory Impact Analysis for Proposed Revisions to the National Oil and Hazardous Substances Pollution Contingency Plan Regulations*, December 3, 2014.

[2] U.S. Environmental Protection Agency, *Guidelines for Preparing Economic Analyses*, 2014.

[3] 5 U.S.C. 601-612

- These principal issues are among many aspects of the analysis that are inconsistent with Executive orders on regulatory review, OMB guidance, and EPA's own guidance for economic analysis.[4]

In the RIA, EPA estimates that the total incremental net present value (at 7 percent) social cost of the rule is $7.8 million, or approximately an annualized cost of $0.7 million over 20 years. In our analysis, we find that the estimated social costs are at least three times higher than EPA's estimate. The cost of the major provisions are given below:

Table ES-1. Estimated Social Costs of the Proposal

	NPV at 7% ($)	Annualized Cost over 20 years at 7% ($)
Product Loss Costs	4,000,000-7,300,000	380,000-720,000
Product Testing Costs	5,300,000	500,000
Stranded Capital/Stockpile Value Loss	1,000,000-1,250,000	90,000-120,000
Label	180,000	20,000
Recovery Costs	1,700,000	160,000
Monitoring Costs	11,000,000	1,000,000
Preauthorization Costs	200,000	20,000
EPA Costs	800,000	80,000
Totals[5]	24,000,000-27,000,000	2,300,000-2,600,000

These costs disproportionately burden small businesses. Most producers in this market are small businesses. The RIA did not find a significant effect on a substantial number of these small businesses. However, when we adjust for the more significant costs in Table ES-1 for the provisions the RIA omits and for real cost of capital, 43 percent of small businesses that the RIA analyzed have compliance costs greater than three percent of their 2011 revenues. As shown in Table ES-2, many small businesses face severe impacts. EPA should evaluate whether it must comply with the procedural and other analytic requirements of the Small Business Regulatory Enforcement Fairness Act (SBREFA).

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[4] Executive Order 12866 Regulatory Review, September 30, 1993.

[5] Throughout this report, totals and summaries may be different due to rounding. Reflecting the likely precision of these estimates, figures are presented to two significant digits or less.

Table ES-2. Economic Impact of the Proposal on Small Businesses

Pct of 2011 Revenue	Number of Product Types	Percent of Companies
<3%	17	57%
3%-5%	5	17%
>5%	8	27%

We recommend that EPA correct these deficiencies including undertaking a comprehensive benefit analysis to determine if the rule is in the national interest. Once EPA conducts its analysis of the incremental net social benefits of each proposed provision, it should select the regulatory options that maximize net social benefits. As EPA's economic analysis guidelines states:

> Preparing high -quality economic analysis can greatly enhance the effectiveness of environmental policy decisions by providing policy makers with the ability to systematically assess the consequences of various actions. An economic analysis can describe the implications of policy alternatives not just in terms of economic efficiency, but also in terms of the magnitude and distribution of an array of impacts.[6]

This report discusses the major issues with the RIA in more detail, provides alternative estimates for the proposal's social costs, and recommends how EPA should correct these flaws before reissuing a proposed rule.

[6] U.S. EPA, *Guidelines*, pg. 1-2.

SECTION 1. INCONSISTENCY BETWEEN BASELINE AND POST-REGULATION MARKET SCENARIOS

1.1. BASELINE IS INCOMPLETE AND INCONSISTENT WITH OMB GUIDANCE

The RIA's description of the analytic baseline is incomplete and inconsistent with OMB guidance.[7] There are several problems with the RIA's regulatory baseline:

- The RIA does not establish baseline conditions for many proposed requirements. As discussed in Section 2 below, the RIA does not fully estimate the social benefits and costs of all of the provisions of the proposal. As it corrects the serious deficiency of omitting cost estimates for proposed provisions, EPA must apply a consistent framework for both baseline and post-proposal market conditions in any final rulemaking.

- The RIA assumes a regulatory baseline where all existing manufacturers must pay to test their products. Even in absence of a rule, manufacturers may choose to conduct additional toxicity testing to demonstrate the inherent low hazard of their product and thus to differentiate their products in the market. Oil spill response organizations and responsible parties may also insist on further toxicity testing from producers due to liability concerns. Other jurisdictions also require testing of oil response products. For example, the corresponding European Union (EU) authority, the European Maritime Safety Agency, lists the dispersant testing requirements as of 2014 in the EU member states.[8] Eight EU members require testing of dispersant products. For all these reasons, producers may have some or all of the proposed testing data already available.

Recommendation. EPA should gather additional information on available data not only to prepare a more accurate economic analysis, but also to properly estimate the incremental social costs of this rule. If additional data already exists, EPA could tailor the final rule's requirements to this data and only mandate additional testing where meaningful data gaps exist.

1.2. PROPOSAL DOES NOT HAVE CONSISTENT SCENARIO FOR THE SOCIAL BENEFITS AND SOCIAL COST ESTIMATES

The RIA assumes all existing products comply with the new testing requirements and remain on the market.[9] The RIA also assumes new products are approved by EPA at the same

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[7] U.S. Office of Management and Budget, Circular A-4 Regulatory Analysis, September 2003.

[8] European Maritime Safety Agency, *Inventory of National Policies Regarding the Use of Oil Spill Dispersants in the EU Member States*, 2014

[9] U.S. Environmental Protection Agency, *Regulatory Impact Analysis for Proposed Revisions to the National Oil and Hazardous Substances Pollution Contingency Plan Regulations*, December 3, 2014.

pace as in the past. In other words, society gains the same benefits from products with the same capabilities and efficacy with or without the rule.

If this post-regulatory scenario is true, the rule has few, if any, social benefits.[10] For example, EPA cites reduced product toxicity on marine life as a potential social benefit. However, any potential ecological harm could only possibly be reduced if some existing (and future) products are withdrawn, not used, or disallowed due to the new toxicity testing requirement and due to the new EPA approval process.

However, the RIA's scenario is not the most plausible post-regulation scenario. It is more likely that some existing products will not remain on the approved list after the rulemaking. This outcome is more likely than the RIA scenario for two reasons. First, EPA may decide not to approve some of the existing products. It seems unlikely that EPA would use its scarce resources to promulgate increased toxicity testing standards if it did not have concerns with the toxicity of some existing products.

Second, given the uncertain sales volume, producers may not make a competitive expected rate of return with the additional testing and application expense and the greater risk of disapproval. In this case, they can be expected not to remain on the schedule after the rule is effective.

There is ample evidence that existing producers face a non-zero risk of product disapproval:

- Interest groups and media alleged significant concerns with oil spill product safety during the 2010 SONS.[11,12] Media attention and interest group attention to the issue does not mean that products are toxic per se; however, the threat of media scrutiny dampens producers' interest in bringing products to market. Increased attention is likely to lengthen EPA's approval process and the costs for the producer to respond to media inquiries. For producers, these response costs decrease the rate of return for oil spill products and potentially raise the costs for other company products.[13]

- Many producers are small businesses and need certain rates of return to compensate for their cost of capital and for the opportunity cost to innovate in

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[10] There is the argument that there is an information externality - the producers of the agents have a more concentrated interest and easier access to the potential toxicity information than the purchasers. However, since the producers are mostly small businesses and the purchasers are typically government agencies or large businesses, the purchasers can be expected to have sufficient resources and potential legal liability to overcome any potential information externality.

[11] See for example, http://www.biologicaldiversity.org/programs/public_lands/energy/dirty_energy_development/oil_and_gas/gulf_oil_spill/dispersants.html.

[12] E. Rothsenthal, *In Gulf of Mexico, Chemicals under Scrutiny*, The New York Times, May 5, 2010.

[13] For example, the parent company, and not individual product, may face public criticism.

other markets. Their internal rate of return requirements may preclude further investments once the costs and the risk increase.

In addition to the effect on existing products, the rulemaking is likely to affect the number of future products. By requiring additional testing and approval, the rulemaking increases market barriers. Some new products may not reach the market in the future due to the costs and risks that new products will not pass the new testing requirements.

1.2.1. Social Cost Estimate

To overcome these limitations, we conduct an alternative estimate to illustrate the importance of considering how the rule will raise market barriers. In Section 6 below, we demonstrate that many smaller producers could face a significant economic impact from the rule, i.e., greater than three percent of their annual revenue. As EPA and Small Business Administration guidance suggests, we assume those firms fail or leave the market due to the regulatory costs.[14]

The social cost is the lost value added to society from these products that leave the market. Value added is the remaining revenue after the costs of the inputs (e.g., labor, materials, and energy) are subtracted. Since value added is often proprietary information and varies from firm to firm, we use a recent value added metric for the chemical industry as a whole. From 2013 Bureau of Economic Analysis data, the average valued added in chemical manufacturing is 44 percent of revenue.[15] We assume that each company in the oil spill product market has a value added of 44 percent of revenue.

Once existing firms leave the market, society loses the value added of their products for Subpart J responses. However, in a competitive marketplace, other firms will enter and offer their own products. Since the regulation is not changing the demand for oil spill response products or acting to reduce the likelihood of spills, post-regulatory supply will rise to meet this unchanged demand. Society, therefore, only temporarily loses the availability of some oil response capabilities once existing producers leave the market in response to regulation.

We assume the value added is lost for three years. While three years is an assumption, it is a reasonable one given the proposed rule's timing. Since existing firms have two years to provide EPA with the new test data, if they decide not to reapply, they will likely try to stay on the market for the full two years. Once EPA removes them from the schedule, competitors will then see a market opening. We assume these competitors offer alternatives and submit their applications to EPA one year later.

We recognize other outcomes are possible. Instead of replacing existing products with new products at a 1:1 ratio, fewer new products could enter the marketplace, leaving the

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[14] U.S. Small Business Administration, *Guide for Government Agencies How to Comply with the Regulatory Flexibility Act*, May 2012. See U.S. EPA, *Final Guidance for EPA Rule writers: Regulatory Flexibility Act as amended by the Small Business Regulatory Enforcement Fairness Act*, November 2006.

[15] U.S. Department of Commerce, Bureau of Economic Analysis, *Value Added by Industry*, November 2014

approved products with a greater market share. In this case, compliance costs would be lower than this estimate; however, an additional social cost would be the loss of resiliency to responds to market disruptions. In addition to small firms, large firms may also not submit the new information due to the expense. On an individual product basis, larger firms may not find it profitable to continue.

Recognizing these limitations, based on these assumptions and the revenue information the RIA provides in Exhibit 7-1, Table 1-1 shows our estimate of the social cost of market disruption/transition due to the proposed rule.

Table 1-1. Social Costs from Market Transition

Annual Revenue	Product Type	Assumed Value Added	Social Cost	Discounted Social Cost (7 percent)
$ 62,000	Solidifiers	$ 27,000.00	$ 81,000.00	$ 74,000.00
$ 160,000	Surface Washing Agent	$ 71,000.00	$ 213,000.00	$ 195,000.00
$ 120,000	Solidifiers	$ 53,000.00	$ 159,000.00	$ 146,000.00
$ 200,000	Solidifiers	$ 88,000.00	$ 264,000.00	$ 242,000.00
$ 30,000	Solidifiers	$ 13,000.00	$ 39,000.00	$ 36,000.00
$ 120,000	Surface Washing Agent	$ 53,000.00	$ 159,000.00	$ 146,000.00
$ 150,000	Surface Washing Agent	$ 66,000.00	$ 198,000.00	$ 181,000.00
$ 1,000,000	Dispersant(2) & Surface Washing Agent	$ 441,000.00	$ 1,323,000.00	$ 1,211,000.00
$ 187,000	Surface Washing Agent	$ 82,000.00	$ 246,000.00	$ 225,000.00
$ 290,000	Bioremediation Agent	$ 128,000.00	$ 384,000.00	$ 352,000.00
$ 280,000	Surface Washing Agent	$ 123,000.00	$ 369,000.00	$ 338,000.00
$ 559,000	Bioremediation Agent	$ 246,000.00	$ 738,000.00	$ 676,000.00
$ 120,000	Bioremediation Agent	$ 53,000.00	$ 159,000.00	$ 146,000.00
Total (rounded)				$ 4,000,000.00

Summary	NPV at 7%	Annualized Cost over 20 years at 7%
Product Loss Costs	$4,000,000	$380,000

The social costs for existing products are estimated to be $4 million. If it takes longer for alternative products to enter the market, these costs would be higher. These social costs do not include the loss of efficacy and resiliency to respond to oil spills during this transition period. In Section 2.3 below, we expand this estimate to add the potential product loss social costs from the proposed ban on products containing NP and NPE.

As discussed previously, the regulation will not only discourage existing producers from re-qualifying, but also discourage new entrants by raising the costs and the risks of market

entry. The RIA assumes new entrants arrive after the rule at the same pace as in the baseline, or 10 per year. This assumption is not supported by economic theory. As discussed below in Section 3.4, we predict that application costs would rise by 40-50 percent for surface washing and stabilizer products, 260 percent for bioremediation products, and 120 percent for dispersants compared to current requirements. While we did not find estimates of the supply elasticity in these markets, such significant increases in prices will affect supply. We assume new product applications decrease by 30 percent from 10 to seven per year in response. The cost implications of this change are further discussed in Section 5.

1.2.2. **Recommendation**

Setting the baseline and policy alternative scenario is one of the key foundations of a regulatory analysis. As EPA's Guidelines for Economic Analysis states: "Because the level of detail presented in the baseline specification is an important determinant of the kinds of analysis that can be conducted on proposed regulatory options, careful thought in specifying the baseline is crucial."[16] The RIA has not shown this careful thought. For the final rule, EPA should understand the available toxicity and efficacy data available from producers. EPA should select, at a minimum, alternative market outcomes from the regulation in which some existing products do not return to the approved list. EPA must estimate the incremental social benefits and social costs from more accurate baselines.

[16] U.S. EPA, *Guidelines*, pg. 5-1.

SECTION 2. SOCIAL COSTS NOT INCLUDED IN THE RIA

2.1. STRANDED PRODUCTION CAPITAL AND PRODUCTS

EPA has not estimated the full social costs from a decision not to approve a product or from a producer's decision not to reapply. For the reasons listed in the previous section, it is plausible that some existing products will not be listed in the future due to the new regulatory requirements. In Section 1.2.1, we estimate the temporary lost value to society. The full social costs include the net change in value of the oil spill response product and the stranded human and physical capital used to produce the product. In addition, existing stockpiles of the dis-approved product can be expected to suddenly have lower value due to the disapproval. While on scene coordinators (OSC) could still allow their use during a spill response, the likelihood is lower once the product is no longer approved.

These costs are potentially significant. Stockpiling response products is an essential strategy to minimize oil spills' consequences. Response products must be deployed within hours of an oil spill; there is not time to ramp up production once a spill happens. Stockpiles are substantial and deployed across the globe. For example, a 2014 EU report lists nearly 3,500 metric tons of more than 40 different compounds in EU member state stockpiles.[17]

EPA has recognized this social cost category in other rulemakings. EPA has banned or phased-out products in other environmental programs. For example, EPA regulates the phase-out of ozone-depleting substances (ODS) under the Clean Air Act (CAA). EPA has conducted several major RIAs for the multiple phase-out decisions for ODS use under Title VI of the CAA. For example, in its Section 604 and Section 606 rulemakings, EPA considered the stranded capital and product costs for the ODS subject to the phase-out. EPA also attempted to estimate the social costs of the reduced efficacy of the substitutes.[18]

2.1.1. Social Cost Estimate

We consider the stranded capital and product costs separately. For the stranded capital, there are several considerations in estimating the social cost. First, the regulation does not destroy the labor and physical capital, but blocks its highest valued use. Workers and entrepreneurs can use their knowledge and machinery to make other products and/or the same product. Second, there is a distinct loss as suppliers move their capital from this highest valued use before the regulation to a lower valued use after the regulation. They also have costs as they retrain and retool their equipment for this shift.

This information guides our estimate. At one extreme, if the regulation completely destroyed the economic value of the enterprise, the loss could be estimated as the enterprise values of the firms. Using a standard financial estimation of small business enterprise value, the loss would be three to five times the firm's intrinsic value (i.e., value added). In this

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[17] European Maritime Safety Agency, *Inventory of National Policies Regarding the Use of Oil Spill Dispersants in the EU Member States*, 2014

[18] For a good summary, see Appendix G of *The Benefits and Costs of the Clean Air Act, 1990 to 2010*, U.S. EPA, 1999.

case, the maximum social cost would be $6 million to $8 million. At the other extreme, in a dynamic economy, excess capital and labor is rapidly moved from one use to another, with almost zero transition cost. Since the regulation fundamentally does not change demand, we assume that the stranded human and physical capital costs will be closer to the low end and assume a loss of $1 million.

For the stranded product costs, knowledge of which existing products will not return to the schedule and the amounts of those products are stockpiled is important information. The RIA gives information about the product types that are likely to leave the market due to regulation (see Section 6 below), but not the specific products.

For current amounts in stockpiles, in support of its 2009 vessel response plan (VRP) rulemaking, the Coast Guard complied data on US stockpiles of products listed on Subpart J.[19] Although this data is comprehensive, it is from 1998. The VRP rulemaking required responsible parties to have contracts with oil spill removal organizations (OSRO) or organic capacity to respond to categorical spills.[20] The rulemaking required owners and operators to have response resources capable of applying dispersants. This rulemaking, and subsequent ones, increased the amount of dispersants stockpiled in the United States.

From public information given by OSROs, most advertised dispersant capacity is for Corexit® formulations. While it is very likely that these organizations also stockpile stabilizers, surface washing agents, and potentially bioremediation agents, we did not find a current, comprehensive list of stockpiled substances.

We therefore constructed our estimate under two scenarios: (1) if EPA does not approve Corexit® formulations; and, (2) If EPA does not approve other agents. For the latter, we include these costs in Section 2.4. A potential ban on Corexit® is discussed in this section.

We gathered available data on current and relatively recent reports of Corexit® formulation stockpiles in the United States and globally. Since the chemical is not approved for use in Europe, we found that global stockpiles are located primarily in the United States and Brazil.[21] Table 2-1 gives the organization and the reported stockpile amount.

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[19] U.S. Coast Guard, "Caps" Review of Oil Spill Response Plan Equipment Capability, at https://homeport.uscg.mil/cgibin/st/portal/uscg_docs/MyCG/Editorial/20080725/combined%20document.pdf?id=213ab594d05ab39ffd49d74a2e188eb8246d23ed&user_id=7ee08b44562829c5918c18d08ac87c29.

[20] 74 FR 45004

[21] http://www.oilspillresponse.com/files/OSRL_GDS_Equipment_20Nov2014.pdf

Table 2-1. Estimated Stockpiles of Corexit® Formulations

Organization	Units	gal/unit	Total
MSRC[22]	1	104,000	104,000
Clean Seas Cooperative[23]	1	20,750	20,750
Oil Spill Response[24]	500	264	132,000
Alyeska Pipeline[25]	1	60,000	60,000
Clean Islands[26]	1	37,260	37,260
LOOP[27]	1	33,600	33,600
Clean Gulf Associates[28]	1	34,650	34,650
CISPRI[29]	1	20,570	20,570
Nalco[30]	1	12,000	12,000

Total (rounded)	450,000
Estimated Value	$ 250,000

[22] https://www-msrc-org-documents.s3.amazonaws.com/major-equipment-list/MSRC_Major_Equipment_List.pdf?AWSAccessKeyId=AKIAIGH7YZWA2HRSZUFA&Expires=1427976589&Signature=lYm5WdnIB7M4XV%2BR1foyl5kUg70%3D

[23] http://www.cleanseas.com/equipment.htm#ABSORBENTS/DISPERSANTS/SPRAY

[24] http://www.oilspillresponse.com/files/OSRL_GDS_Equipment_20Nov2014.pdf

[25] Minerals Management Service, *Assessment of the Use of Dispersants on Oil Spills in California Marine Waters*, S.L. Ross Environmental Research Ltd, June 2002.

[26] http://www.cleanislands.com/

[27] Minerals Management Service, *Assessment of the Use of Dispersants on Oil Spills in California Marine Waters*, S.L. Ross Environmental Research Ltd, June 2002.

[28] http://www.cleangulfassoc.com/equipment.

[29] Minerals Management Service, *Assessment of the Use of Dispersants on Oil Spills in California Marine Waters*, S.L. Ross Environmental Research Ltd, June 2002.

[30] http://www.rrt9.org/external/content/document/2763/904415/1/Nalco%20Dispersant%20Mfgr%20Capability.pdf

Table 2-1 may overestimate the amount stockpiled since different organizations may have overlapping agreements to use the same stockpile. The estimates may also be underestimates since we could not find publicly available data on the stockpiles of some of the larger OSROs such as NRC.[31] Based on a recently reported retail price from an OSRO of $0.55/gallon, the reported stockpiles in Table 2-1 are worth $0.25 million. Since the product is not approved for use in other jurisdictions, if EPA did not approve its continued use, it is likely that current stockpiles will have little residual value.

In summary, the stranded capital/stockpile value loss is the sum of the loss of enterprise value (assumed to be $1 million) and the potential loss of in value of Corexit® formulations. Since the loss in Corexit® formulations is speculative, we present our estimate for this rulemaking outcome as a range.

Summary	NPV at 7%	Annualized Cost over 20 years at 7%
Stranded Capital/Stockpile Value Loss	$ 1,000,000-$1,250,000	$90,000-$120,000

2.1.2. Recommendation

Once EPA establishes a more plausible baseline and post-regulation market conditions, it must account for all social costs that flow from an EPA decision not to approve products currently on the list. These costs include the stranded physical and human production capital, the loss in value of stockpiled products, and any loss in oil spill response efficacy from these decisions. EPA has access to data on oil response product stockpiles that it should use to better estimate the social costs of this rulemaking. EPA's analysis for the final rule should at least be as comprehensive as EPA's approaches in other comparable EPA rulemakings.

2.2. Periodic Revisions of Preauthorization plans

In the proposal, EPA requires Regional Response Teams (RRTs) to review and to update their preauthorization plans for chemical and biological response agents after major oil spills or at least once every five years. The RIA does not estimate the social costs of this proposed requirement.[32]

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[31] http://nrcc.com/our-services/response-services/response-equipment-rental-packages/

[32] On page 37 of the RIA, under the section entitled "Authorization of Agent Use (part 300.910)," the RIA notes that RRTs and Area Committees are required to provide a preauthorization plan and to follow other requirements. The RIA states: "The incremental burden for members of the RRT and Area Committees requires labor hours to participate in the development of preauthorization plans and to add them to RCPS and ACPs." However, it is not clear whether the extremely low burden the RIA uses (2 hours) is to update the 11 or more preauthorization plans or the title of section,

There is evidence that this provision would have meaningful costs if finalized. Recently, EPA's Inspector General (IG) issued a report finding defects in emergency management planning and contingency planning for oil spills.[33] In response, EPA stated:

> Thus, although we generally agree with the recommendations in this report, and they parallel our efforts, two key obstacles (i.e., limited involvement by other agencies and stakeholders and resources) must be overcome for their successful implementation.[34]

The EPA program response to the IG suggests that resources, not interest, is the barrier to updating preauthorization plans. If local governments and other RRT team members are constrained by a lack of resources, then it is not clear how a regulation will alter this reality. The social costs then must reflect this scarcity. The social costs are not limited to the full compensation costs and capital costs necessary to maintain that employee. Unlike private markets, the supply of government officials' time does not increase with greater demand. For this reason, the social costs are the opportunity costs of that official's time - i.e., the social benefits society would gain if the government official spends the time on the next most valuable use of that person's time--enforcement, policy setting, permit approvals, or other public matters. The opportunity cost of this resource - a government agency official's work time - is how EPA's Guidelines for Economic Analysis defines social cost.[35]

2.2.1. **Estimated Social Costs**

Based on the IG's analysis, we estimate the social costs as the resources required to update preauthorization plans more frequently than is current practice. We assume in the baseline that plans are currently revised once every seven years; the proposal requires them every five years or after major oil spills. Based on the IG report finding that some plans had not been updated since the 1990s, a seven year update frequency in the baseline may underestimate social cost. For the post-rule scenario, we use a five year frequency since the other trigger, major spills, are rare and may be expected to lead to preauthorization plan reviews even in the absence of this rulemaking.

The resources required are the time spent revising each plan and the review of each plan by the member organizations. From the agenda of recent RRT meetings, we count the number of distinct Federal and state agencies. In a recent Region 6 RRT meeting, 13 Federal agencies sent staff.[36] We assume that 13 Federal agencies are part of each RRT. States,

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authorization of agent use. In either case, the RIA's estimate is far too low to comply with the proposed requirement.

[33] U.S. Environmental Protection Agency, Office of the Inspector General, *EPA Could Improve Contingency Planning for Oil and Hazardous Substance Response*, February 15, 2013.

[34] Ibid, pg. 19.

[35] "Social cost represents the total burden that a regulation will impose on the economy. It is defined as the sum of all opportunity costs incurred as a result of a regulation where an opportunity cost is the value lost to society of any goods and services that will not be produced and consumed as a result of a regulation," *Guidelines*, p.8-1.

[36] http://www.rrt6.org/Uploads/Files/rrt_semi_annual_meeting_summary_2014_November.pdf

territories, and Tribes also serve on the RRTs. Among the 11 RRTs (one for each EPA region and one for Alaska), we assume 56 States, Tribes, and territories send representatives and participants.

We make some assumptions on the effort needed to revise each preauthorization plan. We assume 20 percent of the RRT's member organizations assign one staff person to serve on an RRT subgroup to revise the plan. Each assigned staff person spends 40 total hours in this revision process. Then, each Federal, State, Tribe, and territory agency reviews the draft revised plan in their organization. Three people in each organization are a part of the review to account for the agency's management and likely legal review. Each person reviewing the draft plan spends six hours reading and discussing the revised plan.

For the costs of government staff time, we use median and mean wages for Federal employees and State employees, respectively.[37] To these wages, we apply recent Federal government values for benefits and overhead to the Federal and State hourly wage values.[38] For the reasons discussed above, this estimate of the opportunity cost of this resource likely underestimates its true social value and thus the social cost of this proposed requirement.

We then compare the social costs for the baseline and proposed requirements - the costs of revising the plans on a seven versus five year frequency. We discount the baseline and proposal time stream of costs at a discount rate of seven percent and take the difference between those time streams as the social cost. The net present value of the additional social cost at a discount rate of seven percent is $0.18 million.

Summary	NPV at 7%	Annualized cost over 20 years at 7%
Preauthorization costs	$180,000	$20,000

2.2.2. Recommendation

The RIA must fully include the social costs and social benefits of this provision and calculate the social costs as the opportunity cost of government agency staff time.

2.3. Prohibition on Chemicals and Proposal to Ban EDCs

The RIA does not estimate the social benefits or the social costs for the proposed option to prohibit approving products containing nonylphenol (NP) and nonylphenol ethoxylates (NPE). More significantly, the RIA did not estimate the social benefits and costs of a regulatory option discussed in the preamble, a ban on endocrine disrupting compounds in approved products.

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[37] U.S. Bureau of Labor Statistics, May 2014 State Occupational Employment and Wage Estimates, http://www.bls.gov/oes/current/oes_ms.htm#11-0000.

[38] See for example, U.S. Office of Personnel Management, Federal Investigative Services Division, *White Paper: Support Services Contract Cost Benefit Analysis.*

Banning any compounds will have social costs and potentially social benefits. As discussed elsewhere in these comments, EPA has acknowledged in other rulemakings the loss of income and loss of productive capital from product bans. These costs must be included in the economic analysis of this option.

One notable option EPA lists is to prohibit materials containing endocrine disrupting compounds from the approved list. More broadly, EPA's proposed prohibition - without any stated criteria for the ban - raises significant market uncertainty. Without transparency into EPA's rationale for the prohibition, producers may decide not to pay for testing costs and risk disapproval, reducing the overall efficacy and variety of products available.

However, the social costs are more significant than the loss of net value in the oil spill response market. Bans by a government agency will cast a cloud over the public's perception of the health and safety of a chemical compound. States and companies maintain lists of "toxic" chemicals and have policies that discourage purchasing and use of chemicals on these lists.[39] By banning a substance in oil spill response products, it is more likely that the market for NP and NPE compounds will shrink throughout the overall economy.

Moreover, the incremental social benefits of the proposed NP and NPE ban are uncertain. The RIA did not quantify these benefits. There is reason to believe that any such social benefits will be small. EPA's own test data provides evidence that the proposed bans would have minimal, if any, social benefits. EPA conducted tests of eight dispersant compound-oil mixtures, some of which contained NPE, for their endocrine activity.[40] EPA found:

> None of the eight dispersants tested displayed biologically significant endocrine disrupting activity, with the exception of a weak response for two of the dispersants (Nokomis 3-F4 and ZI- 400) in one of the tests. This estrogenic result is likely not of biological significance. Cell death (degree the dispersant is toxic to living cells) was observed in some tests at concentrations above 10 parts per million. The endocrine and the cytotoxicity screening were conducted at dispersant concentrations from 0.001 parts per million up to 10,000 parts per million. None of the dispersants triggered cell death at the likely concentrations of dispersants expected in the Gulf.[41]

Therefore, although EPA's study had a small sample set, EPA's test data provides no evidence that these oil spill response compounds with NP and NPE pose any inherent hazard that would justify the social costs of a precautionary ban. The preamble does not provide quantified social benefit estimates to support a ban over other policy alternatives.

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[39] See, for example, Maine's "Chemicals of High Concern," at http://www.maine.gov/dep/safechem/highconcern/.

[40] Judson, R. S., Martin, M. T., Reif, D. M., Houck, K. A., Knudsen, T. B., Rotroff, D. M., Dix, D. J. (2010). *Analysis of Eight Oil Spill Dispersants Using Rapid, In Vitro Tests for Endocrine and Other Biological Activity.* Environmental Science & Technology, 44(15), 5979-5985. doi:10.1021/es102150z

[41] http://www.epa.gov/bpspill/dispersants-qanda.html#toxtest8. See also U.S. Environmental Protection Agency, *Toxicity of Louisiana Sweet Crude Oil (LSC) and chemical dispersed LSC to two Gulf of Mexico aquatic test species,* August 2010.

EPA failed to consider and to evaluate policy alternatives to its proposed ban. Under the Executive orders, EPA must evaluate available policy alternatives and select the ones in which the benefits justify the costs.[42] There are several policy alternatives EPA could choose and could evaluate besides a ban on NP and NPE compounds, such as imposing lesser restrictions or requirements. Such options could tailor the social costs more directly to any potential hazard than a product ban. Only through objective, transparent policy analysis of these alternatives can senior officials and the public weigh the options to decide upon the best policy.

2.3.1. **Estimated Social Costs**

We searched PubMED for published, peer-reviewed articles that found evidence of NP and NPE-containing oil spill response agents. The only study that specifically listed these compounds was EPA's 2010 work in response to the SONS.[43] It identified NP and NPE compounds in two listed dispersants, Nokomis 3-F4 and ZI-400. For the purpose of this estimate, we assume these products are subject to the proposal's ban.

Based on our evaluation, the manufacturers of these products are likely small businesses. To estimate the impact of the ban, we use a similar approach as in Section 1.2.1. The social costs are not permanent; the social value of the banned products is eventually replaced by substitutes or through greater sales of other dispersants. In EPA's testing, the most widely-used dispersant did not contain NP and NPE. The social costs, therefore, are limited to the immediate disruption in the market.

Since we do not know the actual sales of these dispersant products, we average the annual revenue from the three small business manufacturers of dispersant products given in the RIA. As done is Section 1.2.1, we use the value added of the chemical industry to estimate the added value from these products. The social cost of the ban is the lost value from these two products for a three year period.

There are several reasons this approach underestimates the likely social cost. First, medium and large companies may have their products banned also. While these companies may have resources to reformulate their products, there still will be social costs to switch to substitutes. Second, our estimate of two banned products comes from our search of the public, peer-reviewed literature. Given the controversy over dispersants during the SONS and EPA's proposed rule, companies have an incentive not to disclose voluntarily NP and NPE ingredients. Third, it is possible that NP and NPE compounds are used in other product types such as surface washing agents. Fourth, the manufacturers of the two dispersants that EPA identified as containing NP and NPE have similar products on the schedule. If these products also contain NP and NPE, they would also face bans under the proposal.

Finally, we do not estimate the social costs of a more sweeping ban on all EDC compounds. EPA's proposal is too vague for the public to understand its implications. First,

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[42] U.S. Office of Management and Budget, Circular A-4 Regulatory Analysis, September 2003, pg. 16.

[43] Judson RS, Martin MT, Reif DM, et al. Analysis of Eight Oil Spill Dispersants Using Rapid, In Vitro Tests for Endocrine and Other Biological Activity. *Environmental science & technology.* 2010; 44(15):5979-5985. doi:10.1021/es102150z.

EPA does not define what it means to be an EDC. Second, EPA does not state whether all compounded named as an EDC are banned as ingredients or whether *de minimis* amounts are permitted.

Since some of these same products may drop out of the market due to the new testing requirement, we structure our NP product ban cost estimate so as not to double count the product loss estimates already presented. In Section 1.2.1, one product is a dispersant. Our range is simply whether this product's loss is included in the $4 million product loss calculated in Section 1.2.1 or is an additional social cost.

Summary	NPV at 7%	Annualized Cost over 20 years at 7%
Product Loss Costs	$4,000,000-$7,300,000	$380,000-$720,000

2.3.2. Recommendation

The RIA must fully estimate the social costs and the social benefits of the proposal to ban NP and NPE compounds in listed products and the alternative to ban EDCs. To comply with the Executive orders and OMB guidance, EPA must also evaluate policy alternatives to banning compounds and select the one with the greatest net social benefits.

2.4. Expired Product Testing and/or Replacement Costs

EPA proposes that stockpiled products must be tested in order to be used after their expiration date. The RIA does not estimate the social costs of this requirement.

This provision has social costs and potentially social benefits. There are two costs: increased product testing costs and potentially the cost of more frequent product shipments to stockpile locations. The social benefits are potentially more effective products due to this requirement.

2.4.1. Estimated Social Costs

Over time organizations that stockpile oil response products will attempt to minimize the cost of this requirement through inventory management. They can change their ordering procedures to better match inventory to use. More significantly, they can shift sales of products past their official expiration date to other markets. Numerous oil spills and cleanups occur that do not involve Federal OSCs. Since many expiration dates are minimal (e.g., "at least two years"), buyers may be willing to accept response products with older shelf lives for a discount or because their spill response need is less. Vendors/organizations that stockpile will, over time, manage their inventories so that products with young shelf lives are reserved for any Subpart J needs and then, rather than retest to verify their continued use, are migrated to other markets.

This strategy can only work if the major stockpiled products have sufficiently long shelf lives. In the Attachment, the recommended shelf lives for each existing product are listed. We assign a shelf life of 20 years to products reporting unlimited shelf life.

Bioremediation products have the shortest average shelf life of less than three years. Dispersants have the longest average shelf life, with the most widely stockpiled dispersant having an unlimited shelf life. While products vary significantly, there are numerous products in each category with relatively long shelf lives to allow inventory management. To the degree there are effects, management of existing bioremediation products could be most vulnerable to the requirement for additional testing.

For these reasons, we model the social cost as a one-time event to shift management of existing inventories. We assume future purchases and sales will direct products with older shelf lives to cleanups outside National Contingency Plan jurisdiction. We also assume that some operators of existing stockpiles and inventories will decide it is more cost-effective to test rather than replace their existing inventory.[44] They may have quantities of the surface washing products with shorter shelf lives than the average nine year life-span of these products. Different stockpiles will have different products; therefore, the testing decision is based on the number of stockpile-product combinations. We assume that operators find 20 stockpile-product combinations where testing is more cost-effective than selling into other markets or discarding and replacing the product. Since dispersants have a much longer shelf live on average, we only assume four stockpile-product decisions to test. Table 2.2 presents the results based on these assumptions. This provision could have social costs of $1.7 million.

Table 2-2 Estimated Costs of Testing Expired Products

Product	Testing Cost	Number of Stockpiles-Products	Total Cost
Bioremediation Agent	$31,800	20	$1,270,000
Dispersant	$13,050	4	$52,200
Surface Washing Agent	$10,300	20	$206,000
Solidifier	$10,300	20	$206,000
Total			**$1,740,000**

In Section 2.1.1, we consider the social cost of managing stockpiles of existing products that are not approved in the post-regulatory scenario. We assume that OSRO and others manage their inventories of surface washing agents, sorbents, and other products to market them to non-NCP spill responses. In Section 5 we combine all of the provisions that require testing into one estimate so that the analysis makes consistent predictions of market dynamics after the rulemaking.

2.4.2. **Recommendation**

EPA must estimate the social benefits and social costs of this provision if it is included in the final rule. EPA should use current data on stockpile amounts and their expiration dates.

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[44] Or they will replace products in response to the rule up to the expected value of testing them to prove their efficacy. We use the testing cost as the upper limit of the cost to replace existing inventories.

2.5. RECOVERY OF AGENTS FROM ENVIRONMENT

In the proposal, EPA adds a regulatory requirement that responsible parties are obligated to recover certain oil spill response products from the environment. When stabilizers, surface washing agents, and sorbents are used, responsible parties must recover them when it is safe to do so. The RIA does not estimate any social benefits or social costs from this provision.

This requirement has social costs and potentially social benefits. For major spills, it is likely the regulation formalizes the likely site-specific orders by the OSC. By making it a regulatory requirement, the major change is that it subjects the responsible party to the full enforcement authorities under the Clean Water Act. The proposal reduces the government's administrative effort to levy fines and penalties.

The marginal increase in the threat of fines and penalties may change responsible party behavior. Responsible parties may put greater effort into recovery operations and respond to reports of agents in the environment after the spill for a longer period of time. More effort at recovery may then yield some environmental benefits on the margin. However, greater human activity - boats, motors, skimmers - in environmentally-sensitive areas may cause damage while recovering response products. The current framework - to take actions which have the greatest net environmental benefits - is more flexible and allows site-specific balancing of ecological, human safety, and response product use.

2.5.1. Estimated Social Costs

As a practical matter, this provision may not have any additional social costs for major Tier I, Tier II, or Tier III oil spills. On scene coordinators effectively order product recovery for these major spills. For larger spills, the greater level of government oversight in both the baseline and the post-regulation may be expected to lead to similar levels of responsible party activity. For smaller spills, the proposed regulatory requirement could increase social costs by diverting resources into product recovery, especially after the immediate spill response. It is difficult to estimate the additional costs in these site-specific decisions.

Since 2000, there have been on average 13 spills per year of between 5,000-10,000 gallons of oil products into U.S. navigable waters.[45] We assume that these spills are large enough that responsible parties use significant amounts of response products, but not so large that they trigger substantial government oversight.

We assume that the regulatory provision triggers an additional $10,000 in response costs per spill of this medium-to-large spill size. Since daily penalties under the Clean Water Act are up to $37,500 per day per violation, a responsible party may be willing to respond to reports of sorbents and other products in the environment after a spill rather than risk a penalty. They may be willing to spend much more than $10,000 per incident to avoid the fines. However, assuming an average per incident additional cost of $10,000, the net present value (at seven percent) social cost of this requirement is $1.7 million.

[45] United States Coast Guard, *Polluting Incidents In and Around U.S. Waters: A Spill/Release Compendium: 1969 – 2011*, December 2012.

2.5.2. **Recommendation**

EPA should articulate in any final rule the specific social benefits and social costs it believes will occur as a result of this provision. Specifically, EPA should explain the value of this requirement as a regulation as opposed to a site-specific/incident-specific decision that maximizes net environmental benefits.

SECTION 3. UNDERESTIMATED COSTS

3.1. MONITORING COSTS

The proposal requires responsible parties to carry out extensive monitoring of dispersant use, oil concentrations, and ecological effects in the event of a major oil spill where dispersants are used. The RIA estimates the cost of these monitoring requirements to be $500,000 per year for an average of one applicable major, off-shore oil spill per year.

The justification for the cost estimate in the RIA is two studies. However, these studies do not replicate the specific proposed requirements and therefore do not provide adequate justification for the RIA's estimate.

3.1.1. Estimated Social Costs

We build a bottom-up estimate of the social costs to monitor and to report the information listed in the proposed rule. We first make some assumptions concerning the duration of the spill. We assume the responsible party conducts active monitoring of the oil plume and dispersant use for 14 days, followed by 21 days of monitoring of the dissolved oil/dispersant mixture in the water column. Modeling of the oil movement in the water is carried out for 14 days; fluorescence monitoring occurs for the entire 35-day period.

Since dispersants are used, the oil spill is assumed to occur off shore, meaning boats are needed to obtain the water samples and to gather data for the modeling. Background samples are assumed to be gathered at four points upwind and downwind of the plume. In-plume measurements are assumed to be gathered at five points, with three different sampling points in the water column at each point. After the 14 day period, monitoring requirements for the next 21 days fall from five locations in the plume to four areas in the affected area. Background samples are still collected.

Field and test blanks are assumed to be analyzed for each background and plume collection point. The proposal also requires the responsible party to submit a quality assurance plan for the sampling and analysis.

Given the need for rapid decisions, the ecological characterization and risk assessment requirements are divided into two phases. For immediate decisions during the 14 day active response, experts gather available information from already-published studies and regional ecological evaluations. This study also sets the acute toxicity level from available data. Once the response moves to the 21-day monitoring of the ultimate dispersion, a more complete, written ecological characterization report is compiled.

The responsible party assigns 10 managerial and technical staff to manage, carry out, and to report compliance with the monitoring requirements during the 35-day event.

Based on this scenario, we gathered publicly-available price data from OSROs and commercial laboratories to build a social cost estimate. The Oil Spill Response Limited

organization has a global price list and cost data for their Florida-based response location.[46] Table 3-1 gives the principal data used in this cost estimate:

Table 3-1. Prices for Oil Spill Dispersant Use Modeling

Fluorescence Tracking[47]	659	GBP/day
Boat Rental[48]	400	GBP/day
Oil Modeling[49]	2,300	GBP/day
Heavy Metal Analysis[50]	350	$/sample
Volatile Organic Compounds and Benzene, Toluene, Ethylbenzene, and Xylenes[51]	250	$/sample
Total Petroleum Hydrocarbons[52]	150	$/sample
Other Parameters[53]	150	$/sample

Based on this scenario, the estimated monitoring costs are approximately $850,000 per spill. This value is substantially greater than the RIA estimate. This estimate does not include insurance costs, contingency costs, surcharges for rapid delivery, sample shipping costs, travel costs, or other logistical needs. Including these costs would increase the social costs of this provision.

Given the frequency of one spill per year, the net present value and annualized value of these social costs are given below.

Summary	NPV at 7%	Annualized Cost over 20 years at 7%
Monitoring Costs	$11,000,000	$1,000,000

[46] http://www.oilspillresponse.com/

[47] http://www.oilspillresponse.com/files/ScaleOfFees2015.pdf.

[48] Ibid

[49] http://www.oilspillresponse.com/files/ScaleOfFees2015.pdf.

[50] See for example, Price List from Water Testing Labs at http://www.wtlmd.com/wastewater-testing-pricing-maryland-md-va-dc-de.php.

[51] Ibid

[52] Ibid

[53] Ibid

3.1.2. Recommendation

EPA should use the available data from OSRO equipment price lists and laboratory sampling costs to build up a more complete cost estimate for all of the required parameters for dispersant modeling in the proposed rule. With a frequency of dispersant use of approximately once per year, there is sufficient existing data and markets to use market prices to estimate the social cost. EPA should also characterize plume size and expected monitoring requirements from past spill responses. Gathering this data will provide more support for a social cost estimate.

3.2. EPA Costs to Administer the Rule

The RIA underestimates the total resource cost for EPA to administer the rule because it fails to include the total compensation and overhead costs to employ Federal agency staff and it omits the time necessary for EPA to administer the new requirements. The RIA's approach does not comply with EPA's obligations under the Paperwork Reduction Act's implementing regulations (PRA).[54]

The RIA use BLS data from 2011 as estimates of the opportunity cost of government employees. There are several problems with this data:

- It is out of date. The BLS has compensation data available through September 2014.[55] Moreover, the General Services schedule of Federal wages for 2014 is also available as it determines EPA employees' paychecks.

- It underestimates total compensation. It is unclear whether the data the RIA uses is for salary and wages only or for total compensation. It appears that the RIA uses a labor cost of approximately $54 per hour for EPA employees.[56] The

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[54] See 5 CFR 1320.3(a)(1):

(1) *Burden* means the total time, effort, or financial resources expended by persons to generate, maintain, retain, or disclose or provide information to or for a Federal agency, including:

(i) Reviewing instructions;

(ii) Developing, acquiring, installing, and utilizing technology and systems for the purpose of collecting, validating, and verifying information;

(iii) Developing, acquiring, installing, and utilizing technology and systems for the purpose of processing and maintaining information;

(iv) Developing, acquiring, installing, and utilizing technology and systems for the purpose of disclosing and providing information;

(v) Adjusting the existing ways to comply with any previously applicable instructions and requirements;

(vi) Training personnel to be able to respond to a collection of information;

(vii) Searching data sources;

(viii) Completing and reviewing the collection of information; and

(ix) Transmitting, or otherwise disclosing the information.

[55] See the Employment Compensation Survey, http://www.bls.gov/ncs/tables.htm.

[56] Calculated from data in RIA, pg. 36.

RIA should use total compensation measures to account for Federal employee benefits such as pension, health care, disability insurance, and personal leave. This information is readily available in EPA budget documents, in OMB budget memoranda, and from the Office of Personnel and Management.

- The total resource costs for government services is more than the total compensation cost of labor. Government workers do not work in the open air. They need buildings, electricity, computers, telephones, water, and many other overhead costs. Estimating the overhead costs for government employment is a routine practice in Federal budgeting, in program evaluation, and for compliance with the Paperwork Reduction Act.[57]

Further, the RIA does not include the costs for EPA to administer certain provisions of the rule. In addition to the costs to participate in the RRT and AC meetings included in Section 2.2.1, the RIA fails to include the EPA resources necessary to remove a product from the listing schedule.

3.2.1. Estimated Social Costs

We estimate the social costs of EPA resources to oversee the proposed requirements. Although it is likely an underestimate, we retain the RIA's estimate of 26.5 hours for EPA staff to review each product application. Interestingly, the RIA estimates producers of existing products spend only 7.7 to 27.7 hours to submit the required testing data to EPA. Thus, by the RIA, EPA's review time is longer - up to three times longer - than the time needed to prepare the information. If this ratio is correct, EPA's review process could take over 100 hours. We also agree with the RIA's estimate that the same number of hours is needed when a manufacturer submits new testing data after a product has been reformulated.

However, the RIA assumes EPA spends only 2.5 hours reviewing resubmissions of data from applicants. Since EPA has only 90 days to review an application, it is likely to extend its review by asking applicants for additional information. It seems implausible that EPA would need less than 10 percent of its estimated total review time to review resubmitted data. We use an estimate of 20 hours. We also double the amount of time the RIA estimates to review an appeal of a product disapproval. We believe other divisions of EPA, including the Office of General Counsel, will support the Administrator's review of an appeal and a final Agency decision.

For the opportunity cost of EPA personnel time, we use the fully-loaded employment costs. In FY13, the median EPA employee salary was nearly $108,000, or nearly $55 per hour.[58] We apply OPM's benefit and overhead percentages to calculate a fully-loaded rate of

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[57] 5 CFR 1320.3(a)(1).

[58] U.S. Office of Personnel Management, *Salary Information for the Executive Branch Fiscal Year 2013*, April 2014.

$81.50 per hour.[59] This estimate is a slight underestimate of the likely median wage due to general schedule increases since FY13.

Using the post-regulation rate of existing product submissions and new product submissions, we calculate a net present value (at seven percent) of $0.81 million in social costs for EPA to administer the testing requirements.

3.2.2. Recommendation

The RIA should include costs of EPA to carry out all the proposal in its entirety. EPA should include the total compensation cost for government personnel and the overhead costs necessary to house and to equip these employees so that they can perform their new tasks under this proposed regulation. EPA must include these costs and the resources to oversee all of the provisions of the rule in subsequent analyses.

3.3. Testing and Product Listing Costs

The RIA underestimates the costs an organization expends to understand and to comply with this regulation. The RIA assumes essentially that one or two individuals are solitary actors that read, plan, and implement a regulation for a company. In reality, many employees and managers in an organization will be involved in a product's application.

In addition to underestimating the costs for producers, the RIA underestimates the costs to understand, to comply, and to submit product applications. For example, the RIA does not include these costs:

- Laboratory Personnel Understanding the Rule. The RIA does not include the cost for laboratory personnel either within the manufacturer or at a commercial lab to understand the new testing requirements.
- Legal Personnel Time. It seems implausible that a company would file an important submission to EPA without it undergoing legal review. The submission could determine the entire market for a product.
- Response Costs. Often EPA staff will seek clarifications from respondents on their data submissions to the agency. Seeking clarifications in lieu of outright denials/approvals conserves resources for both EPA and respondents. Oddly, the RIA includes EPA's cost to review these re-submissions, but not those of the respondents to prepare them.[60]

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[59] See for example, U.S. Office of Personnel Management, Federal Investigative Services Division, *White Paper: Support Services Contract Cost Benefit Analysis.*

[60] Even if the RIA estimates the full resource cost for the respondent to submit a "perfect" application that does not require clarification, the respondent is not in control of EPA staff comprehension of the submission and their ability to ask questions, triggering respondent costs.

3.3.1. Estimated Social Costs

In Table 3-2 below, we present alternative labor estimates for producers to comply with the rule that addresses the RIA's limitations. We include hours for all the provisions required of producers, including ones the RIA omits like the product label redesign.

Table 3-2. Estimated Skilled Labor Resources Producers Require to Comply with Proposal's Testing Requirements

(Hours)

	Labor Hours Needed		
	Managerial	Technical	Clerical
Hourly Compensation	$ 55.56	$ 55.56	$23.57
Understand Rule	4	8	2
Contract for Tests	2	16	4
Manage Tests	0	8	8
Prepare Documentation	4	40	16
Design Label	0.5	8	8
Legal Review	4	0	2
Revise Documentation	2	4	45
Respond/Follow up	4	16	8
Total (Hours)	20.5	100	93
Total ($) including overhead rate of 17 percent			$10,400

We include hours for essential tasks to prepare the required information. First, the producer must find and contract with an independent certified laboratory for the efficacy and toxicity testing. They must manage the outside laboratory's performance and review the results. In addition, the producer must prepare the written submission to EPA and follow up with any EPA questions.

For labor rates, we use the BLS Employer Cost for Employment Survey data for December 2014.[61] We use the same overhead rate as the RIA, 17 percent, and use the RIA's relative proportion of the application costs among dispersants, solidifiers, surface washing agents, and bioremediation agents. We use these labor costs for product testing in the multiple provisions in the proposal that require product testing.

[61] http://www.bls.gov/ncs/ect/sp/ececqrtn.pdf.

3.3.2. Recommendation

EPA should include all the costs to understand, to explain, and to incorporate the proposed rule into their organizations' operations. EPA is required to perform this analysis under the PRA regulations. In addition, from past applications, EPA should review whether, and if so, the scope of additional information requests it gave to applicants and include these costs in the any final rule. These costs must be assigned to the regulation.

3.4. Limitation To Tested Waters and Efficacy and Toxicity Testing Costs

In the proposal, EPA limits use of a product to the water characteristics of its toxicity and efficacy testing. Product data with just freshwater test data are limited to freshwater uses, for example. The proposed rule expands the number of efficacy tests for bioremediation agents and alters the efficacy test for dispersants. The proposed rule adds new toxicity tests for dispersants and freshwater/saltwater tests for bioremediation, solidifiers, surface washing agents, and other products.

The RIA makes a global assumption that producers chose just one testing environment, salt water or freshwater, and bear the costs for only one test protocol. Rather than chase market demand and maximize net profits, producers instead make their marketing decisions based on the testing cost. The RIA's assumption violates basic economic theory and thus underestimates social costs.

EPA's own data demonstrates this point. Producers list which of the current required toxicity and efficacy testing they have conducted. This information is available on the Subpart J schedule. Except for the bioremediation products, all other products list test data in both the reference saltwater and freshwater species. For bioremediation products, 27 products do not list any information; 21 report test data in both species. In other words, producers currently want to sell in both markets due to the demand for oil spill remediation in both waters. The rulemaking does not change this market demand.

3.4.1. Estimated Social Costs

Since producers currently market to oil spill response demands in both freshwater and saltwater environments, we assume existing and new product manufacturers will pay for the new efficacy and toxicity tests required for both environments. Using the RIA's testing costs, the total laboratory testing cost per product is listed in Table 3-3.

Table 3-3. Testing Costs by Product Type

Dispersants	$13,050
Bioremediation	$31,800
Solidifiers,	$10,300
Surface Washing	$10,300

We use these testing costs to estimate the application cost and the economic impact of the proposed rule. The labor and laboratory testing costs are combined and presented in Section 6.

3.4.2. Recommendation

Consumer demand will drive producers' decisions on marketing, not the testing requirements of the rule. EPA should align its economic analysis to observed producer decisions and thus expect increased testing costs rather than the estimates found in the RIA.

3.5. Retesting Whenever Chemical Composition Changes

In the proposal, producers would be required to retest its products and submit the results to EPA whenever they change the chemical compounds or their concentrations in an approved product. The proposal calls for comment on an option for allowing *de minimis* changes in concentrations or formulations, but proposes retesting for any change. The RIA assigns the same amount of cost to this requirement as it does for a new product application. For the reasons discussed in Section 3.3 above, the RIA underestimates social costs.

3.5.1. Estimated Social Costs

For a cost estimate, we assume that on average one product each year undergoes a change that triggers the requirement to retest. With more than 100 products and product lines on the Subpart J list, the RIA's assumption seems reasonable that approximately one product would undergo a change each year to trigger retesting.

Since we do not know which product type will undergo change, we take an average of the estimated testing costs for the four product types. Therefore, if one product triggers the retesting requirement per year, the net present value (at seven percent) of this proposed provision is $1.7 million/year.

3.5.2. Recommendation

EPA should articulate in any final rule the specific social benefits and social costs it believes will occur as a result of this provision. Specifically, EPA should explain the value of this requirement as opposed to a performance standard that makes manufacturers responsible to demonstrate that formulation changes do not affect toxicity or efficacy.

3.6. Product Label Redesign

In the proposed rule, applicants must submit a sample product label that includes the manufacture and expiration date, the storage conditions, and the mandatory disclaimer language. Producers must not only change the description on the label on the physical product, but also the description on any promotional material, websites, and other presentation forms. The RIA includes the social cost of submitting the sample label in its overall producer cost estimates.

EPA has estimated the social costs of mandatory labeling and labeling changes in multiple rulemakings.[62] The social costs include the one-time labor and capital costs to craft the new label design, any on-going incremental costs to produce the new label, and costs to digitize the information for electronic media.

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[62] See, for example, U.S. Environmental Protection Agency, *Final Technical Support Document, Fuel Economy Labeling of Motor Vehicles: Revisions to Improve Calculation of Fuel Economy Estimates*, EPA420-R-06-017, December 2006.

3.6.1. Estimated Social Costs

We include additional labor hours to account for the changes to a producer's website and marketing materials. Table 3-4 shows the labor costs for this requirement. We add $500 for each producer in capital and material costs to create the new label.

3.6.2. Recommendation

In any final rulemaking RIA, EPA should draw upon other EPA analyses to estimate the labor hours and capital cost requirements to redesign labels and marketing materials.

3.7. Accredited Laboratory Requirement

The proposed rule requires test data be generated from an accredited laboratory. In the baseline, companies could use any laboratory or generate the information in their own laboratories. While the RIA states that it gathered costs from laboratories to conduct the tests, it is unclear whether this cost information is from accredited labs.

3.7.1. Estimated Social Costs

If the RIA data is from accredited labs, then the costs of this requirement is included in the test costs used in the RIA. If it is not, then the RIA's costs for each test is likely underestimated.

3.7.2. Recommendation

In any final rule, the accompanying RIA should disclose the sources of its information so that policy officials and the public can evaluate the incremental social benefits and social costs of this provision as required by the Executive orders on regulatory review.

SECTION 4. THE RIA'S SOCIAL BENEFIT DESCRIPTION IS INACCURATE AND DOES NOT COMPLY WITH EPA'S REGULATORY ANALYSIS REQUIREMENTS

Fundamentally, the RIA fails to comply with the Executive orders on regulatory review because it does not quantify the potential social benefits either in dollars or in any other metric.[63]

Even more troubling, the RIA lists social benefits that are irrelevant to this rule and, if applicable at all, are actually social costs. The RIA lists qualitative benefits from avoiding oil spills such as lost tourism revenue and enjoyment. However, this rulemaking has the potential for aggravating the potential damages from oil spills. To the degree this rulemaking reduces chemical and biological agent availability and innovation, the nation will have less tools available to fight oil spill damage. A possible, but hopefully remote, outcome of this rule is that future oil spill damages would be more severe than they would be if the rule did not occur. Therefore, the potential social costs from oil spills should be considered, if at all, as a social cost of this rulemaking, not a benefit.

The RIA also does not quantify the marginal social benefits from each provision and each regulatory option. Under Circular A-4, agencies are required to estimate the marginal social costs and benefits for each regulatory provision.[64] Since this rulemaking proposes multiple changes to Subpart J, the public and policy officials cannot determine which provisions have net social benefits and which provision provides the most social benefits.

As discussed in Section 1, the other fundamental problem is that the RIA does not have a consistent baseline and post-regulatory alternative to allow a consistent measurement of social benefits. We present our recommended post-regulatory scenario in Section 5.

Recommendation. The RIA does not comply with the basic requirements of a social benefit estimate. First, it must define clearly what the social benefits are from this rulemaking and describe the real social goods that increase from the baseline to the post-regulation scenario. Second, an RIA for any final rulemaking must attempt to quantify the incremental amount of these social goods that can be attributed to this rulemaking and to the specific provisions of the rulemaking that will generate these social goods. Finally, EPA must use the established literature to value the gains, if any, in ecological services from the rulemaking.

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[63] OMB, Circular A-4, pg. 27.

[64] OMB, Circular A-4, pg. 7.

SECTION 5. TOTAL SOCIAL COSTS

5.1. POST-MARKET CONDITIONS

As discussed in Section 1, EPA's economic analysis guidelines emphasize the need for a clear and consistent market baseline and description of post-regulation market conditions. This guidance is especially relevant for this rulemaking because of the interaction of the requirements, the effect on regulated parties' behavior, and the ultimate social cost of the rulemaking.

This section describes a potential sequence of actions flowing from EPA's promulgation of the final rule. The social cost of these actions, and the time they occur, is the structure of our social cost estimates:

- In year 1, EPA promulgates the final rule.
- In year 1 and 2, existing products remain on the market. At the end of year 2, 13 existing products do not submit testing data due to high costs. The value of 13 comes from the results of the economic impact analysis in Section 6. Therefore, the number of existing submitted application is 13 less than the baseline. It is likely that these 13 products include the products banned due to the NP and NPE formulation ban.[65]
- Starting in year 1 and continuing, the rate of new product applications falls from 10 per year to 7 due to increased costs and market barriers
- In year 3, 13 new product applications are submitted to fill the baseline market demand.
- In years 1-3, 64 expired product applications come as suppliers adjust their inventory. We assume 24 in year 1 and 20 each in years 2 and 3.
- One new application appears each year for product reformulation.

[65] In other words, the social costs of the product loss from the banned products are estimated. However, we assume that for the one or two potential products subject to the ban, only one new product is added in the future.

Table 5-1. Pace of Product Applications

	Year						
	1	2	3	4	5	6	7
Baseline							
Existing Products	109						
Annual New Products	10	10	10	10	10	10	10
New Estimate							
Existing Products	59	27					
Expired Products	24	20	20				
Annual New Products/New Formulations	7	8	8	21	8	8	8
Total	**90**	**55**	**28**	**21**	**8**	**8**	**8**

5.2. ADDITIONAL SOCIAL COSTS

5.2.1. Additional Social Costs from this Analysis

Based on the information in Table 5-1, we can integrate the social cost estimates for the different provisions of the rule. Table 5-2 gives the net present value and annualized social costs for the provisions with estimates in this analysis. As in Table 5-1, the social cost for testing includes the costs for expired products, for new products, for retesting of existing products, and for impact of existing products dropping out of the market.

Table 5-2. Additional Social Costs of the Proposal

	NPV at 7% ($)	Annualized Cost over 20 years at 7% ($)
Product Loss Costs	4,000,000-7,300,000	380,000-720,000
Product Testing Costs	5,300,000[66]	500,000
Stranded Capital/Stockpile Value Loss	1,000,000-1,250,000	90,000-120,000
Label	180,000	20,000

[66] This number is the net present value of the weighted annual testing cost multiplied by the number of products per year that is given in Table 5-1. The weighted annual testing cost is that year's mix of dispersant, solidifiers, surface washing, and bioremediation products that submit testing information. For existing products, the each yearly mix of different product types is proportional to the number of each type on Subpart J and the number of each type subtracted/added to the additional testing cost. As discussed in Table 2-2, different product types also have a different frequency of expired product testing. This proportion thus changes the number of each product type tested per year and thus the weighted annual testing cost.

Recovery Costs	1,700,000	160,000
Monitoring Costs	11,000,000	1,000,000
Preauthorization Costs	200,000	20,000
EPA Costs	800,000	80,000

Totals[67]	24,000,000-27,000,000	2,300,000 --2,600,000

5.2.2. Other Costs from the RIA

The RIA included social costs estimates for other proposed requirements that are not re-estimated in this analysis. The RIA estimates the costs for some sorbent manufacturers to submit technical data to be considered on a Sorbent Product List. The RIA estimates the annualized cost to be approximately $15,000. Adding this provision to the costs in Table 5-2 does not materially change the social cost estimate.

[67] Throughout this report, totals and summaries may be different due to rounding. Reflecting the likely precision of these estimates, figures are presented to two significant digits or less.

SECTION 6. THE RIA UNDERESTIMATES THE SMALL BUSINESS IMPACT

The RIA's methodology underestimates the rule's economic impact on small businesses. The proposed rulemaking may have a substantial effect on a significant number of small businesses in the oil spill response product market. EPA should evaluate whether it must comply the procedural and other analytic requirements of the Small Business Regulatory Enforcement Fairness Act (SBREFA).

EPA's methodology underestimates the economic impact in two principal ways. First, as discussed above, it underestimates the direct labor and capital costs to comply with the rule. Second, it underestimates the cost of capital in at least two ways: (1) by assuming owners can finance testing requirements over a longer time period than is likely; and, (2) by underestimating the cost of capital for small businesses. Each of these issues is discussed below.

First, firms producing existing listed products only have two years to reapply and to resubmit the newly-required information. Yet, the RIA annualizes a firm's costs over a 20 year period. It is highly unlikely that a small business could borrow funds over a 20 year period for a product that may only have a limited market for two years. Nor would rational business owners with high opportunity costs for capital tie up their capital for such a long time period. A more realistic financing period is two years.

Second, as EPA's economic guidelines state for economic impact analyses, EPA should use the private cost of capital for small businesses.[68] EPA uses the social cost of capital established by OMB. However, due to the increased loss risk, capital lending institutions or capital investors charge higher effective interest rates for loans to small businesses. The Federal Reserve conducts routine survey of small business lending conditions.[69] In recent months, they have found completed, Small Business Administration-backed small business loans at an interest rate of between 5.1 and 5.6 percent. However, the average effective borrowing rate across all small businesses is much higher. As firms report, if they do not qualify for commercial credit, they use instruments with higher interest rates such as credit cards, trade credit, or debt consolidation. The average effective borrowing rate across all small businesses and all credit needs is substantially greater than seven percent.

We present an alternative estimate of economic impact to small business as an illustration of how large the impacts are likely to be. Table 6-1 shows the testing and the labor costs for existing producers to comply with the proposed rule using different discount rates and annualization periods.

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[68] EPA, *Guidelines*, pg. 9-16.

[69] Federal Reserve Banks of New York, Atlanta, Cleveland and Philadelphia, *Joint Small Business Credit Survey Report, 2014*, February 2015.

Table 6-1. Annualized Compliance Costs for New Testing Requirements, RIA Estimate and Revised Estimate

	Testing Costs	Estimated Labor Costs	RIA's Total Costs	Estimated Total Costs	Annualized at 7% over 20 Years	Annualized at 7% over 5 Years	Annualized at 7% over 2 Years
Dispersants	$13,050	$10,398	$14,240	$23,448	$1,344	$3,473	$12,969
Bioremediation	$31,800	$7,728	$14,042	$39,528	$1,325	$3,425	$21,863
Solidifiers,	$10,300	$6,412	$5,986	$16,712	$565	$1,460	$9,243
Surface Washing	$10,300	$5,077	$5,880	$15,377	$555	$1,434	$8,505

Table 6-2 replicates Exhibit 7-1 of the RIA to present the testing and application costs as a percentage of the firm's annual sales. For economic impact analyses, if a regulation's cost to a firm is three percent of its revenue or more, the general conclusion is that the firm is at risk of closure.[70] The RIA reviewed 30 of the 35 small businesses and found that none exceeded the three percent significant impact threshold. However, using a more realistic financing period and more realistic estimates of costs, 43 percent (13/30) of the firms would face compliance costs greater than three percent of their annual 2011 sales. We recognize that Table 3 uses 2014 values for labor costs and 2011 data for revenue. In any final rule, EPA should update the revenue estimates and recalculate the economic impact using costs and financing options that reflect market conditions.

[70] U.S. Small Business Administration, *Guide for Government Agencies How to Comply with the Regulatory Flexibility Act*, May 2012. See U.S. EPA, *Final Guidance for EPA Rulewriters: Regulatory Flexibility Act as amended by the Small Business Regulatory Enforcement Fairness Act*, November 2006.

Table 6-2. Economic Impact on Small Businesses

Company	Number of Employees	Annual Revenue	Product Type	Annualized Cost	Pct of Revenue
1	1	$ 62,000	Solidifiers	$9,243	14.9%
2	1	$ 1,000,000	Surface Washing Agent	$8,505	0.9%
3	1	$ 160,000	Surface Washing Agent	$8,505	5.3%
4	2	$ 2,500,000	Bioremediation Agent	$21,863	0.9%
5	2	$ 120,000	Solidifiers	$9,243	7.7%
6	2	$ 200,000	Solidifiers	$9,243	4.6%
7	2	$ 30,000	Solidifiers	$9,243	30.8%
8	2	$ 10,000,000	Solidifiers	$9,243	0.1%
9	2	$ 120,000	Surface Washing Agent	$8,505	7.1%
10	3	$ 280,000	Surface Washing Agent	$8,505	3.0%
11	3	$ 1,300,000	Surface Washing Agent	$8,505	0.7%
12	3	$ 150,000	Surface Washing Agent	$8,505	5.7%
13	3	$ 5,000,000	Surface Washing Agent	$8,505	0.2%
14	4	$ 559,000	Bioremediation Agent	$21,863	3.9%
15	4	$ 1,000,000	Dispersant(2) & Surface Washing Agent	$34,442.26	3.4%
16	4	$ 730,000	Solidifiers	$9,243	1.3%
17	4	$ 500,000	Surface Washing Agent	$8,505	1.7%
18	4	$ 187,000	Surface Washing Agent	$8,505	4.5%
19	4	$ 350,000	Surface Washing Agent	$8,505	2.4%
20	5	$ 5,000,000	Bioremediation Agent	$21,863	0.4%
21	5	$ 290,000	Bioremediation Agent	$21,863	7.5%
22	6	$ 120,000	Bioremediation Agent	$21,863	18.2%
23	6	$ 15,000,000	Dispersant(1) & Surface Washing Agent	$21,473	0.1%
24	6	$ 500,000	Surface Washing Agent	$8,505	1.7%
25	6	$ 5,000,000	Surface Washing Agent	$8,505	0.2%
26	7	$ 660,000	Surface Washing Agent	$8,505	1.3%
27	8	$ 3,054,214	Bioremediation Agent	$21,863	0.7%
28	8	$ 2,500,000	Surface Washing Agent	$8,505	0.3%
29	8	$ 1,500,000	Surface Washing Agent	$8,505	0.6%
30	9	$ 2,500,000	Surface Washing Agent	$8,505	0.3%

ATTACHMENT: SHELF LIFE OF LISTED AGENTS[71]

Product	Type	Shelf Life	Assumed Shelf Life (years)
AGROREMED (see SPILLREMED (MARINE)®)	Bioremediation Agent	one year	1
BILGEREMED (see SPILLREMED (MARINE)®)	Bioremediation Agent	one year	1
HYDROREMED (see SPILLREMED (MARINE)®)	Bioremediation Agent	one year	1
INIPOL EAP 22	Bioremediation Agent	one year	1
MUNOX SR®	Bioremediation Agent	over one year when stored at temperatures from 35°F to 95°F	1
OILCLEAN w/ACTIVATOR (see PRO-ACT)	Bioremediation Agent	one year	1
PRISTINE SEA II	Bioremediation Agent	As a dry bacterial blend, the shelf life is 1 year, and as a liquid bacterial mixture, the shelf life is 6 months	1
PRO-ACT	Bioremediation Agent	One year	1
SPILLREMED (INDUSTRIAL)	Bioremediation Agent	one year	1
VAPORREMED (see SPILLREMED (MARINE)®)	Bioremediation Agent	one year	1
BILGE CLEAR (see S-200)	Bioremediation Agent	1½ years if kept in the unopened original container	1.5
NANOBITE (see S-200)	Bioremediation Agent	1½ years	1.5
OIL GONE EASY (see S-200)	Bioremediation Agent	1½ years if kept in the unopened original container	1.5
S-200	Bioremediation Agent	1½ years	1.5
S-200C (see S-200)	Bioremediation Agent	1½ years	1.5

[71] Information compiled from individual products listed on schedule as of March, 2015. http://www2.epa.gov/emergency-response/alphabetical-list-ncp-product-schedule-products-available-use-during-oil-spill.

S-200 OIL CLEAN (see S-200)	Bioremediation Agent	1½ years	1.5
S-200 OIL GONE (see S-200)	Bioremediation Agent	1½ years	1.5
SHEENCLEAN (see S-200)	Bioremediation Agent	1½ years	1.5
Z-11 (see S-200)	Bioremediation Agent	1½ years	1.5
BIO-REGEN HYDROCARBON (see SOIL RX)	Bioremediation Agent	Shelf life exceeds 2 years when properly stored in the unopened original container	2
BIOREM-2000 OIL DIGESTER™	Bioremediation Agent	Two years when stored within the storage temperature range in the original container	2
BIOREM-2000 SC (see BIOREM-2000 OIL DIGESTER™)	Bioremediation Agent	Two years when stored within the storage temperature range in the original container	2
JE1058BS	Bioremediation Agent	2 years if stored in a cool dry area	2
REMEDIADE™	Bioremediation Agent	unopened is 2 years	2
SOIL RX	Bioremediation Agent	two years	2
SP 7010 (see REMEDIADE™)	Bioremediation Agent	two years	2
SUMP SAFE BIO-RECLAIM	Bioremediation Agent	The material must be used within 24-48 hours of constitution of the powdered bacteria portion in water. The dry material must be kept at -20°C and is stable for two years at this temperature. Once sent to the field, the material may be stored on ice for up to two weeks prior to constitution	2
SYSTEM E.T. 20 (formerly MCW.B 20)	Bioremediation Agent	2 years if maintained at 4ºC (39ºF)	2
WASTE AWAY®	Bioremediation Agent	Two years	2
WMI-2000	Bioremediation Agent	2 years if stored at ambient temperatures	2
B&S INDUSTRIAL (see STEP ONE)	Bioremediation Agent	Over 3 years	3
BET BIOPETRO (formerly BET BIOPETRO HEAVY)	Bioremediation Agent	More than 3 years in unopened original shipping container, stored in cool dry area	3
BIOWORLD BIOREMEDIATION HYDROCARBON TREATMENT PRODUCTS	Bioremediation Agent	exceeds 3 years when stored properly in cool, dry place out of direct sun light	3
SHAMANTRA BIO (see SHAMANTRA GREEN)	Bioremediation Agent	3 years when stored in sealed silos, polydrums, poly bags or totes	3
SHAMANTRA GREEN	Bioremediation Agent	3 years when stored in sealed silos, polydrums, poly bags or totes	3
STEP ONE	Bioremediation Agent	Over 3 years	3

VB591™, VB997™, BINUTRIX®	Bioremediation Agent	three years	3
ACT TERRA FIRMA	Bioremediation Agent	If kept in a sealed container with desiccant bag the shelf life is 5 years	5
ACT-TF (see ACT TERRA FIRMA)	Bioremediation Agent	If kept in a sealed container with desiccant bag the shelf life is 5 years	5
DRYLET™ MB BIOREMEDIATION	Bioremediation Agent	Minimum 5 years, with proper storage, in original packaging. Freezingdoes not harm shelf life; however, extreme heat (over 180°F) for long periods of time can shorten shelf life	5
DUALZORB®	Bioremediation Agent	unopened product is 5 years if not exposed toweather or direct sunlight	5
ERGOFIT MICRO MIX AQUA	Bioremediation Agent	5 years	5
LAND AND SEA RESTORATION	Bioremediation Agent	More than 5 years in unopened original shipping container, store in a cool, dry place	5
MICROSORB SC (see OPPENHEIMER FORMULA)	Bioremediation Agent	5 years	5
OIL SPILL EATER II	Bioremediation Agent	5 years	5
OPPENHEIMER FORMULA	Bioremediation Agent	5 years	5
The OPPENHEIMER FORMULA I (see OPPENHEIMER FORMULA)	Bioremediation Agent	5 year	5
TRAILZORB (see DUALZORB®)	Bioremediation Agent	5 years if not exposed to weather or direct sunlight	5
WHITZORB (see DUALZORB®)	Bioremediation Agent	unopened product is 5 years if not exposed to weather ordirect sunlight	5
MICRO-BLAZE®	Bioremediation Agent	Minimum 10 years	10
SUPERSPERSE™ WAO2500	Dispersant	one year	1
MARINE D-BLUE CLEAN™	Dispersant	2 years, if unopened	2
NOKOMIS 3-AA	Dispersant	two years	2
FFT-SOLUTION™	Dispersant	5 years	5
NEOS AB3000	Dispersant	five years	5
MARE CLEAN 200 (formerly MARE CLEAN 505)	Dispersant	10 years when stored indoors	10
NOKOMIS 3-F4	Dispersant	15 years or more if stored correctly in plastic drums	15
ACCELL CLEAN® DWD	Dispersant	unopened drums of ACCELL CLEAN® DWD is unlimited	20

BIODISPERS (formerly PETROBIODISPERS)	Dispersant	unlimited	20
COREXIT® EC9500A (formerly COREXIT 9500)	Dispersant	The shelf life of unopened drums of COREXIT® EC9500A is unlimited	20
COREXIT® EC9500B	Dispersant	The shelf life of unopened drums of COREXIT® EC9500B is potentially unlimited when containers remain capped and sealed to prevent contamination and evaporation of solvents	20
COREXIT® EC9527A (formerly COREXIT 9527)	Dispersant	The shelf life of unopened drums of COREXIT® EC9580A is unlimited	20
DISPERSIT SPC 1000™	Dispersant	The shelf life of Dispersit SPC 1000™ is unlimited in unopened containers	20
FINASOL® OSR 52	Dispersant	unlimited when the containers remain capped and sealed	20
JD-109	Dispersant	unlimited	20
JD-2000™	Dispersant	unlimited	20
SAF-RON GOLD	Dispersant	unlimited in unopened containers	20
SEA BRAT #4	Dispersant	Indefinite when stored properly	20
SEACARE ECOSPERSE 52 (see FINASOL® OSR 52)	Dispersant	potentially unlimited when the containers remain capped and sealed to prevent contamination and evaporation of solvents, and are stored in a place with stable temperatures and are unexposed to direct sunlight	20
SEACARE E.P.A. (see DISPERSIT SPC 1000™)	Dispersant	unlimited in unopened containers	20
SF-GOLD DISPERSANT (see SAF-RON GOLD)	Dispersant	unlimited	20
ZI-400	Dispersant	Unlimited in sealed polydrums of totes	20
ZI-400 OIL SPILL DISPERSANT (see ZI-400)	Dispersant	Unlimited in sealed polydrums of totes	20
MARI-ZYME (see ZYME-FLOW)	Miscellaneous Oil Spill Control Agent – MOSCA	1 year minimum	1
UNITED 658 PETRO-ZYME (see ZYME-FLOW)	Miscellaneous Oil Spill Control Agent – MOSCA	1 year minimum	1
ZYME-FLOW	Miscellaneous Oil Spill Control Agent – MOSCA	1 year minimum	1
ZYME-TREAT (see ZYME-FLOW)	Miscellaneous Oil Spill Control Agent – MOSCA	1 year minimum	1
PX 700™	Miscellaneous Oil Spill Control Agent – MOSCA	Two years	2

NORSOREX® APX	Miscellaneous Oil Spill Control Agent – MOSCA	3 years from date of production	3
ALSOCUP	Miscellaneous Oil Spill Control Agent – MOSCA	five years	5
CIAGENT (formerly CI AGENT, CHEAP INSURANCE, and PETRO-CAPTURE)	Miscellaneous Oil Spill Control Agent – MOSCA	5 years, if stored in cool dry area, away from direct sunlight	5
ELASTOL	Miscellaneous Oil Spill Control Agent – MOSCA	5 years when stored at temperatures below 150ºF	5
LIQUID ELASTOL (see ELASTOL)	Miscellaneous Oil Spill Control Agent – MOSCA	5 years when stored at temperatures below 150ºF	5
OIL BOND®	Miscellaneous Oil Spill Control Agent – MOSCA	5 years	5
OPFLEX®	Miscellaneous Oil Spill Control Agent – MOSCA	5 years	5
SEPARATE (see ELASTOL)	Miscellaneous Oil Spill Control Agent – MOSCA	5 years	5
PES-51	Miscellaneous Oil Spill Control Agent – MOSCA	6 years (unopened drum), 1 year (opened drum)	6
GELCO 200	Miscellaneous Oil Spill Control Agent – MOSCA	10 years when stored in dry, cool area out of sunlight	10
AQUA N-CAP™ POLYMER (see OIL SOLUTIONS POWDER)	Miscellaneous Oil Spill Control Agent – MOSCA	Not Limited	20
OIL SOLUTIONS POWDER	Miscellaneous Oil Spill Control Agent – MOSCA	Not limited	20
RAPIDGRAB 2000™	Miscellaneous Oil Spill Control Agent – MOSCA	unlimited	20
WASTE-SET #3200®	Miscellaneous Oil Spill Control Agent – MOSCA	unlimited	20
WASTE-SET #3400®	Miscellaneous Oil Spill Control Agent – MOSCA	unlimited	20
SILTECH OP-40	Surface Collecting Agent	three years	3
THICKSLICK 6535	Surface Collecting Agent	three years	3
CLEAN SPLIT (see SPLIT DECISION SC)	Surface Washing Agent	one year	1

DUO-SPLIT (see SPLIT DECISION SC)	Surface Washing Agent	one year	1
GREEN TECHNOLOGIES SOLUTIONS-OIL RECOVERY (GTS-OR)	Surface Washing Agent	One year minimum when stored between 50°F and 100°F	1
SPLIT DECISION SC (formerly SPLIT DECISION)	Surface Washing Agent	one year	1
TXCHEM HE-1000™	Surface Washing Agent	One year in a sealed container	1
AQUACLEAN	Surface Washing Agent	18 months if stored between 50ºF and 104ºF, and away from acids	1.5
BIOGRASS® EXTRA	Surface Washing Agent	Two years minimum	2
CN-110	Surface Washing Agent	2 years	2
CORIBA 700 ER (see CORIBA 700 SR)	Surface Washing Agent	Two years unopened minimum	2
CORIBA 700 OS (see CORIBA 700 SR)	Surface Washing Agent	Two years unopened minimum	2
CORIBA 700 SR	Surface Washing Agent	Two years unopened minimum	2
CORIBA 713 ER (see CORIBA 713 SR)	Surface Washing Agent	Two years unopened minimum	2
CORIBA 713 OS (see CORIBA 713 SR)	Surface Washing Agent	Two years unopened minimum	2
CORIBA 713 SR	Surface Washing Agent	Two years unopened minimum	2
DE-SOLV-IT CLEAN AWAY APC SUPER CONCENTRATE	Surface Washing Agent	Two years in sealed drums or totes	2
DE-SOLV-IT INDUSTRIAL FORMULA	Surface Washing Agent	Two years in sealed polydrums or totes	2
DO-ALL #18	Surface Washing Agent	at least two years	2
ECOVOOM-MARINE (see JEP-MARINE CLEAN)	Surface Washing Agent	minimum 2 years	2
ETHOS CLEAN	Surface Washing Agent	2 years in a sealed (unopened) container (tote, drum, pail) as delivered. 1 year in an opened container).	2
JEP-MARINE CLEAN	Surface Washing Agent	minimum 2 years	2
MARINE GREEN CLEAN™	Surface Washing Agent	2 years	2
MARINE GREEN CLEAN PLUS™	Surface Washing Agent	2 years	2

NONTOX™	Surface Washing Agent	minimum of two years	2
OSR-10	Surface Washing Agent	2 years in a sealed (unopened) container (tote, drum, pail)	2
PROCLEANS	Surface Washing Agent	2 years at recommended temperatures if unopened	2
SOC 10	Surface Washing Agent	two years	2
SUPERALL #38 (see TOPSALL #30)	Surface Washing Agent	two years	2
TOPSALL #30	Surface Washing Agent	two years	2
TULXA	Surface Washing Agent	Two years if stored in dry, fresh environment	2
EO ALL PURPOSE SOAP-LAVENDER	Surface Washing Agent	Three years	3
FIREMAN'S BRAND SPILLCLEAN (see SPILLCLEAN)	Surface Washing Agent	3 years. Avoid extreme heat and store in a dry, cool area	3
SPILLCLEAN or SPILLCLEAN ["Concentrate"]	Surface Washing Agent	three years	3
CLEAN GREEN	Surface Washing Agent	5 years in sealed polydrums or totes	5
CLEANGREEN® PLANET WASH (see CLEAN GREEN)	Surface Washing Agent	5 years in sealed polydrums or totes	5
MICRO CLEAN (see NATURE'S WAY HS)	Surface Washing Agent	five years	5
NATURE'S WAY HS	Surface Washing Agent	five years	5
NATURE'S WAY PC (see NATURE'S WAY HS)	Surface Washing Agent	five years	5
POWERCLEAN (see NATURE'S WAY HS)	Surface Washing Agent	five years	5
PREMIER 99	Surface Washing Agent	5 years	5
SAFE KLEEN	Surface Washing Agent	Five years if stored in tightly closed containers under dry conditions within temperature range	5
SC-1000™	Surface Washing Agent	Minimum of 5 years	5
SIMPLE GREEN®2013 Reformulation	Surface Washing Agent	five years	5
G-CLEAN OSC-1809	Surface Washing Agent	Five to ten years in sealed polydrums or totes	7.5
OIL SPILL CLEANUP (see G-CLEAN OSC-1809)	Surface Washing Agent	Five to ten years in sealed polydrums or totes	7.5
DYNAMIC GREEN™	Surface Washing Agent	6-10 years in sealed poly drums or totes	8

BIOSOLVE® HYDROCARBON MITIGATION™ AGENT	Surface Washing Agent	10+ year shelf life if unopened	10
CYTOSOL	Surface Washing Agent	Closed container: 10 years in a dry environment. Open container: 1 year in a warm, humid environment. The product does not deteriorate appreciably over time, but wil grow bacteria if water condensation accumulates in the container.	10
F-500	Surface Washing Agent	15 years when stored between 35ºF - 130ºF in unopened containers	15
NOKOMIS 5-W	Surface Washing Agent	15 years or more if stored correctly in plastic drums	15
ACCELL CLEAN® SWA	Surface Washing Agent	unopened drums of ACCELL CLEAN® SWA is unlimited	20
ALL PURPOSE CLEANER & REMEDIATOR (see GREEN BEAST™ OIL SPILL & ODOR REMEDIATOR)	Surface Washing Agent	Unlimited in sealed polydrums or totes	20
AWAN PRA OIL FIELD SOLUTION™ (see EPA OIL FIELD SOLUTION™)	Surface Washing Agent	Unlimited in sealed polydrums or totes (as delivered). Avoid direct sunlight.	20
BG-CLEAN™ 401	Surface Washing Agent	unlimited	20
COREXIT® EC9580A (formerly COREXIT 9580 SHORELINE CLEANER)	Surface Washing Agent	unlimited	20
ENVIROCLEAN (formerly ENVIRO CLEAN 165)	Surface Washing Agent	Unlimited if unopened	20
ENVIRONMENTAL 1 CRUDE OIL CLEANER	Surface Washing Agent	Unlimited in sealed polydrums or totes	20
ENVIRONMENTAL 1 WASHING AGENT (see ENVIRONMENTAL 1 CRUDE OIL CLEANER)	Surface Washing Agent	Unlimited in sealed polydrums or totes	20
EPA OIL FIELD SOLUTION™	Surface Washing Agent	Unlimited in sealed polydrums or totes (as delivered). Avoid direct sunlight.	20
E-SAFE©	Surface Washing Agent	Unlimited if left in unopened containers stored at 40º - 110ºF and away from direct sunlight	20
GLOBAL ENVIRONMENTAL CLEANER™ (see EPA OIL FIELD SOLUTION™)	Surface Washing Agent	Unlimited in sealed polydrums or totes	20
GOLD CREW SW	Surface Washing Agent	20 years (unopened)	20

GREEN BEAST OIL SPILL & ODOR REMEDIATOR	Surface Washing Agent	Unlimited in sealed polydrums or totes	20
GREEN BEAST WASHING AGENT (see GREEN BEAST OIL SPILL & ODOR REMEDIATOR	Surface Washing Agent	Unlimited in sealed polydrums or totes	20
HYDRO-CLEAN™ (see EPA OIL FIELD SOLUTION™)	Surface Washing Agent	Unlimited in sealed polydrums or totes	20
NALE-IT	Surface Washing Agent	Indefinite	20
NATURAMA G3 A-5	Surface Washing Agent	unlimited	20
PETRO-CLEAN	Surface Washing Agent	Indefinite when stored properly	20
PETRO-GREEN ADP-7	Surface Washing Agent	indefinite	20
PETROMAX PSC 3	Surface Washing Agent	Unlimited when stored in unopened, sealed 5 gallon containers, 55 gallon drums, polydrums, and totes	20
PETROMAX SOIL CLEANING AND WASHING AGENT (see PETROMAX PSC 3)	Surface Washing Agent	Unlimited when stored in unopened, sealed 5 gallon containers, 55 gallon drums, polydrums, and totes	20
PETROTECH 25	Surface Washing Agent	unlimited shelf-life when maintained in the factory sealed containers and stored within the prescribed temperature limits	20
SANDKLENE 950	Surface Washing Agent	Unlimited in sealed drums or totes	20
SHEEN-MAGIC©	Surface Washing Agent	Unlimited if left in unopened containers stored at 40º - 110ºF and away from direct sunlight	20
SIMPLE GREEN®	Surface Washing Agent	unlimited	20
VERU-SOLVE™ MARINE 200 HP	Surface Washing Agent	Indefinite, product will not degrade over time	20

To: Conmy, Robyn[Conmy.Robyn@epa.gov]
Cc: Wilson, Gregory[Wilson.Gregory@epa.gov]
From: Principe, Vanessa
Sent: Mon 1/25/2016 2:17:00 PM
Subject: RE: BSEE wave tank dispersion report comments
E12PG00037_Draft Final Report__sept2015_OEM.DOCX
RE: BSEE funded wave tank study of oil dispersion that was conducted by EPA, DFO Canada, NJIT, Dalhousie and USF.

Hey Lady:

No worries, I loose stuff all the time! Here it is (both the document and the email trail just in case). Home today, and perhaps a couple more days because of the snow…it was a lot of snow for us. I'll share some pics in a bit.

Please call if you have any questions, you know where to find us.

Vanessa – 301-262-0311 (home); 301-509-3428 (mobile)

Greg – 202-351-1357

Be safe, V

From: Conmy, Robyn
Sent: Friday, January 22, 2016 10:10 AM
To: Wilson, Gregory <Wilson.Gregory@epa.gov>; Principe, Vanessa <Principe.Vanessa@epa.gov>
Subject: BSEE wave tank dispersion report comments

I am working on the reviews of the report and cannot find the one from OEM??? I have the simulants one, but not the wave tank report. I know you submitted one b/c we spoke on the phone about some of the comments. I hate to ask, but can you resend?

Thanks,

Robyn

Snowing yet? Are you blizzard prepared? People have lost their dang minds here and we are forecasted for less than 5 in. Empty grocery store shelves, long lines for gas. I don't get it. It must be mayhem in DC. Stay warm and safe…. and have fun snowball fights!

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Proposed Changes to Subpart J of the National Contingency Plan

The U.S. Environmental Protection Agency (EPA) is proposing to amend Subpart J of the National Contingency Plan (NCP) requirements for the use of dispersants, and other chemical and biological agents to respond to oil spills in waters of the U.S. The public has the opportunity to comment on this proposed rule during a 60-day period following its publication in the *Federal Register.*

Background

In 1994, EPA revised the NCP in response to the passage of the Oil Pollution Act of 1990. The final rule established the current Subpart J regulatory requirements with respect to authorization of use, data requirements, dispersant effectiveness and toxicity testing protocols, surface washing agent testing protocol, bioremediation agent testing protocols, and requirements for adding products to the Schedule.

Why is EPA proposing these amendments?

In April 2010, the Deepwater Horizon underwater oil well blowout released significant quantities oil into the Gulf of Mexico. Dispersants were deployed over several miles of the Gulf both applied on surface slicks and directly injected into the oil gushing from the well riser, totaling close to two million gallons of dispersants. This unprecedented use of dispersants raised many questions about efficacy, toxicity, environmental impacts and monitoring challenges that EPA now seeks to address through these proposed revisions to Subpart J..

These proposed changes would help to ensure that chemical and biological agents have met rigorous efficacy and toxicity requirements, and that product manufacturers have provided important use and safety information. This would equip the planning and response community with the proper information to evaluate the use of products to effectively mitigate health and environmental effects from oil discharges from occurring in navigable waters and other areas of concern.

What changes are proposed?

The proposed rule would amend Subpart J requirements for the use of dispersants, and other chemical and biological agents to respond to oil spills and considers concerns that arose during the Deepwater Horizon response regarding toxicity, efficacy, long-term environmental impacts, endangered species protection, and human health. Additionally, revisions are proposed to area planning requirements for dispersant use authorization and to toxicity and efficacy testing methodologies and listing thresholds, and advanced monitoring techniques are proposed. The revisions are a major component of EPA's effort to inform the use of dispersants and other chemical or biological agents, to respond to oil spills based on the federal government's experiences in the Gulf. Specifically, the rule proposes changes to:

- *Definitions.* Revise definitions for Bioremediation agents, Burning agents, Chemical agents, Dispersants, Sinking agents, Sorbents, and Surface washing agents; new definitions for Bioaccumulation, Bioconcentration, Biodegradation, Biological agents, Herding agents, Products, and Solidifiers; and remove the definitions for Miscellaneous Oil

Spill Control Agent (MOSCA) and Surface collecting agents.

- *General Requirements.* Revise to reflect new and amended regulatory definitions.
- *Authorization of Use.* Revise to clarify planning and preauthorization responsibilities, establish limitations and prohibitions on the use of certain agents, establish requirements for storage and use of agents, clarify authorities for requiring supplemental testing and information on agents, establish requirements for agent recovery from the environment, and provide notification of agent use.
- *Monitoring Requirements for Agent Use.* New monitoring requirements for major and/or continuous discharges.
- *General Product Information for Schedule Listing.* Revise and add new requirements, including designation of and testing for all product categories under which the listing is requested, Material Safety Data Sheets, sample product labels, shelf life, collection and recovery, persistence in the environment, storage and use conditions, physical and chemical properties, component identities, concentration limits on National Water Quality Standards and Criteria contaminants, laboratory accreditations, submission of all testing data and calculations, production capabilities, and any other data or certification informing the product's performance capabilities or environmental benefits.
- *Dispersant Testing and Listing Requirements.* New efficacy testing methodology using a baffled-flask test, new developmental and sub-chronic toxicity testing requirements, revisions to acute toxicity testing methodologies, and revisions to the listing criteria, and new use limitations to saltwater environments.
- *Surface Washing Agent Testing and Listing Requirements.* Revise the acute toxicity testing methodology and listing requirements, new efficacy testing requirements and listing criteria, and new use limitations based on product testing for salt and/or freshwater environments.
- *Bioremediation Testing and Listing Requirements.* Revise the efficacy and acute toxicity testing methodologies and listing criteria, new exceptions for specified products, and new use limitations based on product testing for salt and/or freshwater environments.
- *Solidifier Testing and Listing Requirements.* Revise the acute toxicity testing methodology and listing criteria, and new use limitations based on product testing for salt and/or freshwater environments.
- *Herding Agent Testing and Listing Requirements.* Revise the acute toxicity testing methodology and listing criteria, and new use limitations based on product testing for salt and/or freshwater environments.
- *Sorbent Testing and Listing Requirements.* Publicly available list of known, non-proprietary sorbents to be made available in lieu of listing sorbents on the Schedule; requirements for data, information and certifications for sorbent products with components other than those specifically identified as non-proprietary.
- *Submissions of Confidential Business Information.* Revise the allowable confidential business information claims and reporting procedures.
- *Addition of a Product to the Schedule.* Revise submission requirements, including the package contents, EPA's review, request for review of decision, changes to a listed product, and transitioning of listed products from the current Schedule to the new Schedule.
- *Mandatory Product Disclaimer.* Revise the product disclaimer requirements.

- *Removal of a Product from the Schedule.* Revise basis for removal and appeals process.
- *Appendix E to Part 300 – Oil Spill Response.* Remove this appendix to the NCP.

How can you comment on the proposal?

Comments must be received on or before [Insert date 60 days after publication in the Federal Register]. Comments must be identified by Docket ID No. EPA-HQ-OPA-2006-0090, and submitted by one of the following methods:

- Federal Rulemaking Portal: http://www.regulations.gov. Follow the on-line instructions for submitting comments.
- Mail: The mailing address of the docket for this rulemaking is EPA Docket Center (EPA/DC), Docket ID No. EPA-HQ-OPA-2006-0090, 1200 Pennsylvania Avenue, NW., Washington, DC 20460.
- Hand Delivery: Such deliveries are only accepted during the Docket's normal hours of operation, and special arrangements should be made for deliveries of boxed information. The Public Reading Room is open from 8:30 a.m. to 4:30 p.m., Monday through Friday, excluding legal holidays. The telephone number for the Public Reading Room is 202-566-1744.

For More Information

Read the Proposed Subpart J Rule Amendment:
http://www.epa.gov/emergencies/

Visit the EPA Office of Emergency Management Web site:
http://www.epa.gov/emergencies

Call the National Contingency Plan Product Schedule Information Line
(202) 260-2342 (phone)

To Report an Oil or Chemical Spill

Call the National Response Center:
(800) 424-8802 or (202) 267-2675
TDD (202) 267-4477

EPA NEWS ADVISORY

Proposed Amendments to Subpart J Product Schedule of the National Oil and Hazardous Substances Pollution Contingency Plan

Contact: Stacy Kika

The U.S. Environmental Protection Agency (EPA) is proposing to amend requirements under Subpart J of the National Oil and Hazardous Substances Pollution Contingency Plan (NCP) to address concerns raised by the regulated community and response officials, and to incorporate scientific advances and lessons learned from oil discharges, particularly the recent Deepwater Horizon event.

Specifically, EPA is proposing to amend the Subpart J Product Schedule regulatory requirements for a submitter to have chemical, biological, or other spill mitigating substances listed on the Product Schedule, to revise the efficacy and toxicity testing protocols, to add new listing criteria, and to require additional health and safety information. Additionally, EPA is proposing amendments to the area planning requirements for dispersant use authorization, and new monitoring techniques for certain major discharges, including data reporting. EPA is also proposing new evaluation criteria for removing products from the Schedule. These changes would ensure that the emergency response community has the information it needs to make informed spill response decisions in the field to protect human health and the environment.

More information about the proposed NCP Subpart J Product Schedule Rule is available at http://www/epa/gov/emergencies.

6560-50-P

ENVIRONMENTAL PROTECTION AGENCY

40 CFR Parts 110 and 300

[EPA-HQ-OPA-2006-0090]; [FRL-]

RIN 2050-AE87

National Oil and Hazardous Substances Pollution Contingency Plan

AGENCY: U.S. Environmental Protection Agency (EPA)

ACTION: Proposed Rule

SUMMARY: The U.S. Environmental Protection Agency (EPA or the Agency) proposes to amend the requirements in Subpart J of the National Oil and Hazardous Substances Pollution Contingency Plan (NCP) that govern the use of dispersants and other oil spill mitigating devices and substances when responding to oil discharges into waters of the U.S. The proposal addresses the efficacy, toxicity, environmental fate and monitoring of dispersants, and other chemical and biological agents, as well as public, state, local, and federal official's concerns on their authorization and use. Specifically, the proposal amends the Subpart J regulatory requirements for the NCP Product Schedule (Schedule) by adding new listing criteria, revising the efficacy and toxicity testing protocols, and clarifying the evaluation criteria for removing products from the Schedule. The Agency also proposes amended requirements for the authorities, notifications, monitoring and data reporting when using chemical or biological agents in response to oil discharges in waters of the U.S. These requirements are anticipated to encourage the development of safer and more effective spill mitigating products, and would better target the use of these products to reduce the risks to human health and the environment. Further, the amendments are intended to ensure that On-Scene Coordinators (OSCs), Regional Response Teams (RRTs), and Area Committees have sufficient

information to support agent preauthorization or authorization of use decisions.

DATES: Comments must be received on or before **[Insert date 60 days after publication in the Federal Register].**

ADDRESSES: Submit your comments, identified by Docket ID No. EPA-HQ-OPA-2006-0090, by one of the following methods:

- Federal Rulemaking Portal: *http://www.regulations.gov*. Following the on-line instructions for submitting comments.
- Mail: The mailing address of the docket for this rulemaking is EPA Docket Center (EPA/DC), Docket ID No. EPA-HQ-OPA-2006-0090, 1200 Pennsylvania Avenue, NW., Washington, DC 20460.
- Hand Delivery: Such deliveries are only accepted during the Docket's normal hours of operation, and special arrangements should be made for deliveries of boxed information.

Instructions: Direct your comments to Docket ID No. EPA-HQ-OPA-2006-0090. EPA's policy is that all comments received will be included in the public docket without change and may be made available online at *http://www.regulations.gov*, including any personal information provided, unless the comment includes information claimed to be Confidential Business Information (CBI) or other information whose disclosure is restricted by statute. Do not submit information that you consider to be CBI or otherwise protected through *http://www.regulations.gov*. The *http://www.regulations.gov* website is an "anonymous access" system, which means EPA will not know your identity or contact information unless you provide it in the body of your comment. If you submit an electronic comment, EPA recommends that you include your name and other contact information in the body of the comment and with any disk or CD-ROM you submit. If EPA cannot read your comment due to technical difficulties and cannot contact you for clarification, EPA may not be able to consider

your comment. Electronic files should avoid the use of special characters, any form of encryption, and be free of any defects or viruses. Comments and suggestions regarding the scope of any future rulemaking should be clearly differentiated from comments specific to today's proposal (e.g., label Suggestions for Future Rulemaking and Comments on Current Proposal).

Docket: All documents in the docket are listed in the *http://www.regulations.gov* index. Although listed in the index, some information is not publicly available, e.g., CBI or other information whose disclosure is restricted by statute. Certain other material, such as copyrighted material, will be publicly available only in hard copy. Publicly available docket materials are available either electronically in *http://www.regulations.gov* or in hard copy at the EPA Docket, EPA/DC, EPA West, Room 3334, 1301 Constitution Avenue, NW. Washington, DC. The Public Reading Room is open from 8:30 a.m. to 4:30 p.m., Monday through Friday, excluding legal holidays. The telephone number for the Public Reading Room is 202-566-1744 to make an appointment to view the docket.

FOR FURTHER INFORMATION CONTACT: For general information, contact the Superfund, TRI, EPCRA, RMP, and Oil Information Center at 800–424–9346 or TDD at 800–553–7672 (hearing impaired). In the Washington, DC metropolitan area, contact the Superfund, TRI, EPCRA, RMP, and Oil Information Center at 703–412–9810 or TDD 703–412–3323. For more detailed information on specific aspects of this proposed rule, contact William Nichols at 202-564-1970 (nichols.nick@epa.gov), U.S. Environmental Protection Agency, Office of Emergency Management, Regulation and Policy Development Division, 1200 Pennsylvania Avenue, NW, Washington, DC 20460–0002, Mail Code 5104A, or visit the Office of Emergency Management website at http://www.epa.gov/oem/.

SUPPLEMENTARY INFORMATION: The contents of this preamble are:

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III. Statutory Authority and Delegation of Authority

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1. Definitions

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d. Bioremediation Testing and Listing Requirements

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f. Herding Agent Testing and Listing Requirements

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VI. Summary of Proposed Rule Provisions

VII. Statutory and Executive Order Reviews

A. Executive Order 12866: Significant Regulatory Action

B. Paperwork Reduction Act

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E. Executive Order 13132: Federalism

F. Executive Order 13175: Consultation and Coordination with Indian Tribal Governments

G. Executive Order 13045: Protection of Children from Environmental Health Risks and Safety Risks

H. Executive Order 13211: Actions That Significantly Affect Energy Supply, Distribution or Use

I. National Technology Transfer and Advancement Act

J. Executive Order 12898: Environmental Justice

I. General Information

In April 2010, the Deepwater Horizon underwater oil well blowout released significant quantities of oil into the Gulf of Mexico. The blowout released oil from one mile below the sea surface. Approximately one million gallons of dispersants over a three-month period were

deployed on surface slicks over thousands of square miles of the Gulf, and approximately one quarter of a million gallons of dispersants were, for the first time, injected directly into the oil gushing from the well riser. This unprecedented use of dispersants raised many questions about efficacy, toxicity, environmental trade-offs and monitoring challenges that EPA now seeks to address through these proposed revisions to Subpart J.

Today's proposed revisions to Subpart J address the use of dispersants, and other chemical and biological agents to respond to oil discharges into waters of the U.S. by enhancing consideration of the toxicity, efficacy, long-term environmental impacts, endangered species protection, and human health concerns raised during the Deepwater Horizon spill response. Additionally, area planning requirements for dispersant use authorization, toxicity thresholds and advanced monitoring techniques are also proposed. The proposed amendments are a major component of EPA's effort to inform the use of dispersants and other chemical or biological agents when responding to oil discharges, based on lessons learned from the federal government's experiences in the Gulf.

The proposed amendments would help to ensure that only products that perform effectively in laboratory testing would be listed on the NCP Product Schedule (Schedule) for use in mitigating the effects of oil spills in the environment. Manufacturers would be required to provide more detailed product application materials, ecological toxicity data, and human health and safety information, including more detailed instructions for product application in the field. Prohibitions on using products in certain areas under certain conditions determined by On-Scene Coordinators (OSCs), RRTs, and EPA would help ensure that first responders are better able to prevent spilled oil from reaching navigable waters and other areas of concern, and to mitigate environmental effects when spills occur. The required submission of additional product toxicity information would aid

OSCs and RRTs when evaluating specific product information and when deciding whether and which products to use to mitigate hazards to the environment and human health caused by discharges or threatened discharges of oil.

Specifically in today's action, the Agency proposes to amend:

- *Definitions.* Amended definitions for Bioremediation agents, Burning agents, Chemical agents, Dispersants, Sinking agents, Sorbents, and Surface washing agents; new definitions for Bioaccumulation, Bioconcentration, Biodegradation, Biological agents, Fertilizers, Herding agents, Products, and Solidifiers; and removed definitions for Miscellaneous Oil Spill Control Agents (MOSCA) and Surface collecting agents.
- *General Requirements.* Revisions to reflect new and amended regulatory definitions.
- *Authorization of Use.* Revisions to clarify planning and preauthorization responsibilities, establish limitations and prohibitions on the use of certain agents, establish requirements for storage and use of agents, clarify authorities for requiring supplemental testing and information on agents, establish requirements for agent recovery from the environment, and provide notification of agent use.
- *Monitoring Requirements for Agent Use.* New monitoring requirements for major and/or continuous discharges.
- *General Product Information for Schedule Listing.* Revisions and new requirements, including designation of and testing for all product categories under which the listing is requested, Material Safety Data Sheets, sample product labels, shelf life, collection and recovery, persistence in the environment, storage and use conditions, physical and chemical properties, component identities, concentration limits on National Water

Quality Standards contaminants, laboratory accreditations, submission of all testing data and calculations, production capabilities, and any other data or certification informing the product's performance capabilities or environmental benefits.

- *Dispersant Testing and Listing Requirements.* New efficacy testing methodology using a baffled flask test, new developmental and sub-chronic toxicity testing requirements, revisions to acute toxicity testing methodologies, revisions to the listing criteria, and new use limitations to saltwater environments.
- *Surface Washing Agent Testing and Listing Requirements.* Revisions to the acute toxicity testing methodology and listing requirements, new efficacy testing requirements and listing criteria, and new use limitations based on product testing for salt and/or freshwater environments.
- *Bioremediation Agent Testing and Listing Requirements.* Revisions to the efficacy and acute toxicity testing methodologies and listing criteria, new exceptions for specified products, and new use limitations based on product testing for salt and/or freshwater environments.
- *Solidifier and Herding Agent Testing and Listing Requirements.* Revisions to the acute toxicity testing methodology and listing criteria, and new use limitations based on product testing for salt and/or freshwater environments.
- *Sorbent Requirements.* Publicly available list of known, non-proprietary sorbents to be made available in lieu of listing sorbents on the Schedule; requirements for data, information and certifications for sorbent products with components other than those specifically identified as non-proprietary.

- *Submissions of Confidential Business Information.* Revisions to the allowable confidential business information claims and reporting procedures.
- *Addition of a Product to the Schedule.* Revisions to submission requirements, including the package contents, EPA's review of submission package, request for review of decision, changes to a listed product, and transitioning of listed products from the current Schedule to the new Schedule.
- *Mandatory Product Disclaimer.* Revisions to the product disclaimer requirements.
- *Removal of a Product from the Schedule.* Revisions to basis for removal and appeals process.

EPA estimates industry may incur total costs that range from approximately $360,000 to $504,000 annually. The benefits of this action are estimated to range from approximately $2.7 million to $3.8 million annually. The net benefits of the proposed rule are estimated to range from $2.3 million to $3.3 million annually. This action does not pose significant impacts on a substantial number of small entities. The Regulatory Impact analysis, which can be found in the docket, provides more detail of the cost and benefits methodology.

COST AND BENEFITS OF THE PROPOSED RULE

	Annualized Benefits and Costs, 20 years	
	Annualized at 3%	Annualized at 7%
Costs	$360,409	$504,326
Benefits	$2,688,268	$3,775,717
Net Benefits (Benefits –Costs)	$2,365,957	$3,323,087

II. Entities Potentially Affected by This Proposed Rule

Industrial Category	NAICS Code

Chemical Manufacturing	325
Merchant Wholesalers, Nondurable Goods	424
Professional, Scientific, and Technical Services	541
Waste Management and Remediation Services	562
Oil and Gas Extraction	211

The list of potentially affected entities in the above table includes manufacturers and users of chemical and biological agents, and other oil spill mitigating devices and substances used as countermeasures against oil discharges. The Agency's goal is to provide a guide for readers to consider regarding entities that potentially could be affected by this action. However, this action may affect other entities not listed in this table. If you have questions regarding the applicability of this action to a particular entity, consult the person(s) listed in the preceding section entitled **FOR FURTHER INFORMATION CONTACT**.

III. Statutory Authority and Delegation of Authority

Under sections 311(d) and 311(j) of the Clean Water Act (CWA), as amended by section 4201 of the Oil Pollution Act of 1990 (OPA), Pub. L. 101-380, the President is directed to prepare and publish the National Oil and Hazardous Substances Pollution Contingency Plan (NCP) for removal of oil and hazardous substances. Specifically, section 311(d)(2)(G) requires the President to include a schedule identifying "dispersants, other chemicals, and other spill mitigating devices and substances, if any, that may be used in carrying out" the NCP. The authority of the President to implement this portion of the CWA is currently delegated to EPA in Executive Order 12777 (56 FR 54757, October 22, 1991). Subpart J of the NCP governs the use of dispersants, and any other chemical or biological agent to respond to oil discharges (40 CFR part 300 series 900).

IV. Background

A. Historical Background

The Council on Environmental Quality (CEQ) first published the National Oil and Hazardous Materials Pollution Contingency Plan in 1970 (35 FR 8508). Among its elements was Annex X – Schedule of Dispersants and other Chemicals to Treat Oil Spills. Annex X provided a basic regulatory framework that included authorization of use, restrictions, and information requirements to be submitted to the Federal Water Quality Administration (FWQA). This initial schedule advocated mechanical and other control methods, and the removal and proper disposal of oil from the environment. It also specified that dispersants may be used in accordance with the schedule if other control methods are judged to be inadequate or infeasible, and if certain information requirement and usage requirements were met. As a listing requirement, manufacturers had to provide the FWQA with methods for analyzing the chemical components in fresh and salt water, or reasons why such analytical methods could not be provided. Except to prevent or substantially reduce hazard to human life or limb, or substantial hazard of fire to property, dispersants were restricted from use on or in any: distillate fuel oil; spill of oil less than 200 barrels in quantity; shoreline; waters less than 100 feet deep; waters containing major populations or breeding or passage for species of fish or marine life which may be damaged or rendered commercially less marketable by exposure to the dispersant or dispersed oil; waters where the winds and/or currents are of such velocity and direction that the dispersed oil mixtures would likely, in the judgment of the FWQA, be carried to shore areas within 24 hours; and waters where such use may affect surface water supplies.

The CEQ revised the National Oil and Hazardous Materials Pollution Contingency Plan, renaming it the National Oil and Hazardous Substances Pollution Contingency Plan (NCP) in 1971 (36 FR 16215). The NCP was amended in 1971 (36 FR 18411) and 1972 (37 FR 28208), with no substantive changes to Annex X.

As a result of the Federal Water Pollution Control Act (Public Law 92-500) of 1972, CEQ again revised the NCP, including revisions to Annex X (38 FR 21887, August 13, 1973). The title of Annex X changed to the "*Schedule of Chemicals to Remove Oil & Hazardous Substances Discharges.*" While the Schedule still advocated mechanical and other control methods, and the removal and proper disposal of oil from the environment, it broadened its applicability to chemical agents, including dispersants. It also recognized separate authorizations of use for chemical agents on minor, medium and major discharges. In addition, the revised schedule required an official report from a recognized laboratory with a description of the analytical methods employed and results obtained in determining the chemical and biological characteristics of the chemical agent, but no longer required the submission of those analytical methods. Biological and burning agents were not part of the Schedule, which expressed caution on their use.

In 1975, CEQ again revised the NCP, including Annex X (40 FR 6282). Annex X was now the "*Schedule of Chemicals and Other Additives to Remove Oil & Hazardous Substances Discharges*," and it was "...revised and expanded to provide more precise and definitive information, concerning substances which may be employed to remove discharges." Additionally, Executive Order 11735 (38 FR 21243, August 3, 1973) made EPA responsible for Annex X. Chemical agents or any other substance not specifically defined in the Schedule were to be considered for use on a case-by-case basis. The Schedule advocated the development and utilization of mechanical control methods to remove or mitigate oil, and to remove, mitigate, or neutralize hazardous substances discharges from the environment, with subsequent proper disposal. The Schedule intended for no harmful quantities of any substance were to be applied to waters to remove or mitigate the effects of oil or hazardous substances discharges. The Schedule also provided procedures for authorization of use for different categories. Separate regulatory

authorization of use for dispersants or other chemical agents was established for minor, medium, and major discharges. Product shelf life, toxicity and effectiveness, and analytical methods needed to obtain such data were among the technical data requirements. Similar provisions were tailored to surface collecting agents and biological additives.

In 1982 EPA amended the NCP; the revision included rewriting of Annex X as Subpart H of the revised Plan (47 FR 31180). The Agency allowed OSCs to authorize the use of dispersants or other chemicals to treat discharges of oil, provided they were listed in the previous Annex X, with the following limitations:

- Authorization applies only to discharges of oil, not to releases of hazardous substances.
- OSCs may only authorize the use of dispersant or other chemicals on EPA's Acceptance list, which included the twenty-eight products tested and found acceptable for their intended purpose in the previous Annex X.
- State consultation provisions regarding the use of any dispersant or other chemicals in its waters required the OSC to obtain concurrence from the EPA representative to the RRT.

The new Subpart H provided for a case-by-case authorization by the EPA Administrator or designee of the use of dispersants or other chemicals not on EPA's Acceptance list in treating oil discharges or hazardous substances releases. However, EPA did not include testing procedures or a process for designation of dispersants or other chemicals as acceptable for use.

In 1984, EPA published amendments to Subpart H, including adding Appendix C (49 FR 29192). The amendments specified testing and data requirements for listing of dispersants, surface collecting agents, or biological additives on the Schedule to ensure sufficient data was available for the OSC. Standardized testing procedures generated comparable data regarding the products' effectiveness and toxicity. Appendix C detailed the methods and types of apparatus to be used in

carrying out the revised standard dispersant effectiveness and toxicity tests, as well as the format required for summary presentation of product test data. Listing of a product on the Schedule was neither a recommendation nor authorization for use on an oil discharge, but rather a confirmation that data submission requirements had been satisfied. EPA's listing on the Schedule did not confirm its safety or effectiveness, however, or constitute an endorsement; a new requirement was established that a written disclaimer to this effect, or EPA's written disclaimer, be included in all product technical literature or advertisements. Products previously listed under Annex X were included on the new Schedule as the previous data requirements were sufficient to permit OSCs to make informed decisions about product use

The amendments also provided for OSC authorization of use for burning agents on a case-by-case basis with the concurrence of the EPA RRT representative and the States, and prohibited the use of sinking agents in waters of the United States. They encouraged advance planning by allowing the OSC to act without the concurrence of the RRT and affected States if these parties have approved a plan identifying the products that may be used in specific contexts. The amendments also allowed the OSC to authorize the use of any product (including products not on the Schedule) without obtaining the concurrence of the EPA representative to the RRT or the affected States if the use of a dispersant, surface collecting agent, or biological additive is necessary to prevent or substantially reduce hazard to human life, and there is not sufficient time to obtain concurrence. However, the OSC was to inform the EPA RRT representative and the affected States of the use of a product as soon as possible, and obtain their concurrences for the product's continued use in the situation once the threat to human life had subsided. These provisions were designed to eliminate delays in life threatening situations, such as spills of highly flammable petroleum products in harbors or near inhabited areas.

Subpart H was re-designated to Subpart J with minor changes in 1990 (55 FR 8666), with those definitions present in Subpart H moved to Subpart A, and a new definition and data requirements for miscellaneous spill control agents added.

The Schedule listed mostly dispersants until 1994 when, in response to the passage of the Oil Pollution Act in 1990, EPA revised the NCP (59 FR 47384). The final rule significantly revised Subpart J to its current regulatory requirements with respect to authorization of use, data requirements, dispersant effectiveness and toxicity testing protocols, surface washing agent toxicity testing protocol, bioremediation agent effectiveness testing protocol, and requirements for adding products to the Schedule. As a result, several dispersants were re-classified as surface washing agents because they did not pass the dispersant efficacy test threshold, but were effective in removing oil from solid surfaces.

B. Current Statute and Regulation

Section 300.910 of Subpart J addresses the authorization of the use of products on the NCP Product Schedule and specifies the conditions under which OSCs may authorize the use of dispersants, other chemicals, and other spill mitigating devices and substances. Sections 300.915 and 300.920 describe the data requirements and the process for adding products to the Product Schedule. To list a product on the Schedule, Subpart J currently requires the submission of technical data on the product to EPA. Data on dispersants, surface collecting agents, surface washing agents, and miscellaneous oil spill control agents must include the results of the toxicity test set for these products in Appendix C of the NCP. Data on dispersants must also include the results of the dispersant effectiveness test, while bioremediation agents must include results of the bioremediation effectiveness test, also set forth in Appendix C. These tests are conducted at the expense of the product manufacturers and must be performed by laboratories experienced with

Appendix C protocols.

The raw data and a summary of the results from these tests are then submitted to EPA, where they are reviewed to confirm that the data are complete and that the specified procedures were followed. The data requirements for placement of a product on the Product Schedule are designed to provide sufficient data for the OSCs to judge whether and in what quantities a product may be used to control a particular discharge. Inclusion of a product on the Product Schedule means only that the data submission requirements have been satisfied. The listing of a product on the Schedule *does not* mean that the product is recommended or authorized for use on an oil discharge *nor* does it imply that EPA has confirmed the safety or effectiveness of the product or in any other way endorsed the product for the use listed or for other uses. The standardized testing procedures set forth in Appendix C provide OSCs comparable data regarding the toxicity, effectiveness, and other characteristics of different products. Since the 1994 revisions of the Subpart J requirements, XX products have been added to the Product Schedule, bringing the total number of products on the Schedule to XX (as of the date of this Notice).

C. Advanced Response Planning

RRTs and the Area Committees are required to address, as part of their planning activities, the desirability of using appropriate dispersants, other chemical or biological agents, or other spill mitigating devices. The RRTs and Area Committees generally develop "pre-authorization plans" which address the specific context in which products can be used under OSC direction, but without obtaining specific concurrences from other RRT members. Preauthorization does not mean that an OSC has blanket authority to use a product without any notification to the RRT. The use of chemical or biological agents is only one approach of many available, such as mechanical collection or *in-situ* burning, and decisions about their use should be carefully weighed against

environmental considerations.

Area Committee responsibilities include enhancing contingency planning and ensuring preplanning of joint federal, state, and local response efforts, and expediting decisions on the use of dispersants and other spill mitigating devices and substances. The ACPs must include a list of the equipment, dispersants or other spill mitigating devices and substances, and personnel available to ensure an effective and immediate removal of a discharge and to ensure mitigation or prevention of a substantial threat of a discharge, and a description of the procedures to be followed for obtaining an expedited decision regarding the use of dispersants, and the means to monitor use of chemical countermeasures. Presently, coastal ACPs contain some type of pre-approval zones for dispersants, while most Regional Contingency Plans (RCPs) address other product categories, such as bioremediation and surface washing agents. RRTs, in cooperation with the states and federal agencies, have addressed the requirements for the conduct of in-situ burns (ISB) of oil discharges in their RCPs.

To facilitate the best possible response, it is important that the regional-level and area-level contingency planning efforts of the RRTs and Area Committees, respectively, are coordinated closely with each other and are consistent. Section 300.910(a) authorized the RRTs to review and either approve, disapprove, or approve with modification the preauthorization plans developed by Area Committees. This advanced planning has allowed the OSC to authorize the use of products without obtaining the specific concurrences, if the RRT representatives from EPA and the states with jurisdiction over waters to which a preauthorization plan applies, and DOC and DOI natural resource trustees approve their use in advance. The OSC primarily uses the Schedule to confirm if a product is listed, analyze toxicity and efficacy data, note worker health and safety precautions, understand proper product application, and compare one product to another in order to make the

most informed decision on how to mitigate an oil discharge.

This planning has contributed to more authorization of chemical agent use on oil discharges. These agents have increasingly been considered and used in the field, as evidenced by the response to the Deepwater Horizon oil spill, increases in research and case studies presented at national and international oil spill conferences, increases in research and development funding from private and government stakeholders, RRT efforts to plan for their use, and increases in requests for EPA experts to advise field personnel on the use of such products. While many research efforts have helped to clarify issues and added information on the toxicity, efficacy, proper use, and human health impacts of these response technologies, uncertainties still arise. The Agency has an interest in resolving the issues that arise from the use of chemical and biological agents in spill responses, as its representatives are asked to concur with chemical and biological agent use for marine and freshwater spills.

The use of non-mechanical oil spill mitigating technologies, such as dispersants, surface washing agents, sorbents, solidifiers, bioremediation agents, and others has prompted debate and actions among the oil response community throughout the United States and the world. The National Academy of Sciences report titled "Oil Spill Dispersants: Efficacy and Effects" (2005), recommends steps to be taken to better support policymakers and spill responders faced with making choices regarding the use of dispersants as part of spill contingency planning efforts or during actual spills. The Agency has collaborated with industry and other federal agencies on the "Selection Guide for Oil Spill Applied Technologies (Selection Guide)"[1] designed to simplify the evaluation of such chemical agents with a step-by-step process for determining which categories of technologies, and which specific products and strategies, might be useful in various oil spill

[1] The Selection Guide can be found on the National Oceanic and Atmospheric Administration's (NOAA) Office of Response and Restoration website at http://response.restoration.noaa.gov/topic.

situations. The United States Coast Guard has promulgated new requirements for certain ocean going vessels to have only Schedule listed dispersant response capabilities while operating in designated dispersant pre-approval zones (74 FR 45004, August 31, 2009).

Today's proposed amendments are aimed at ensuring that chemical and biological agents have met rigorous efficacy and toxicity requirements, that product manufacturers provide important use and safety information, and that the planning and response community is equipped with the proper information to authorize and use the products in a judicious and effective manner.

V. This Action

This proposal addresses the efficacy, toxicity, environmental fate and monitoring of dispersants, other chemical and biological agents, and spill mitigating devices, as well as public, state, local, and federal officials concerns on their authorization and use. The sections below explain the proposed revisions by section.

A. Discharge of Oil

The Agency is proposing changes to §110.4 that comport with the revised definitions for chemical and biological agents used in Subpart J. The language in §110.4 is currently specific to dispersants and emulsifiers. This proposal replaces the terms "dispersants and emulsifiers" with the broader term "chemical or biological agent" as proposed to be amended in 300.5. The proposed definition for chemical agents includes elements, compounds, or mixtures designed to facilitate the removal of oil from a contaminated environment and mitigate any deleterious effects. The proposed definition for biological agents includes microorganisms (typically bacteria, fungi, or algae) or biological catalysts, such as enzymes, able to enhance the biodegradation of a contaminated environment. Chemical and biological agents would include dispersants and emulsifiers that are cited in §110.4. By revising 40 CFR Part 110, the Agency is clarifying that any

chemical or biological agent added to a discharge of oil with the intent to circumvent any provision of 40 CFR Part 110 is prohibited. To further reflect this proposed revision, the Agency is amending the section title to "Chemical or Biological Agents." EPA believes the proposed amendment is consistent with USCG regulations, which prohibits the discharge of chemicals or other substances into the sea that circumvent discharge conditions specified in their regulation (33 CFR 151.10(g)). EPA requests comment on these revisions.

B. Subpart A – Introduction

1. Definitions

The Agency is proposing to amend the definitions for Bioremediation agents, Burning agents, Chemical agents, Dispersants, Sinking agents, Sorbents, and Surface washing agents. The Agency is also proposing new definitions for Bioaccumulation, Bioconcentration, Biodegradation, Biological agents, Bioremediation, Fertilizers, Herding agents, Products, and Solidifiers. Finally, the Agency is removing the definition for Miscellaneous Oil Spill Control Agent (MOSCA) and Surface collecting agents.

a. Revised Definitions:

Bioremediation agent - The Agency is proposing to revise the definition to read: "Bioremediation agents are biologically derived compounds such as enzymes, nutrients, or microorganisms deliberately introduced into a contaminated environment to increase the rate of biodegradation and mitigate any deleterious effects caused by the contaminant constituents. Bioremediation agent categories include, but are not limited to, biological agents, enzymes, and fertilizers containing bio-available forms of nitrogen and phosphorus." This definition expands on the current definition by adding specific examples of bioremediation agents. This clarification will help manufacturers of products in identifying the type of product, and hence, what testing

requirement they will need to comply with to have a product listed on the Schedule.

Burning agents - The Agency is proposing to revise the definition to read: "Burning agents are additives that improve the combustibility of the materials to which they are applied through physical or chemical means." Burning agents may include inorganic compounds in the form of gelling agents such as aluminum soap. For example, the fuel used in helitorch systems is a mixture of powdered gelling agents with either gasoline, jet fuel, or a diesel/gas mixture, which are organic compounds.[2] The Agency believes both the inorganic gelling agent and the organic fuel (e.g., gasoline) meet the burning agent definition by improving the combustibility of the materials to which they are applied through physical or chemical means. The Agency considered including ignition devices in the definition of burning agent because improving the combustibility of oil and igniting that oil could be considered one and the same. The intent would be to clarify the potential that not only substances, but also the devices holding those substances, be considered in the case-by-case authorization of use of such agents. The Agency rejected this approach since many devices either deliver the agent to the oil to be burned and do not enter the water, or are consumed by the burning along with the agent. In addition, the Agency is not proposing a generic listing for these agents at this time because they are expected to burn off during use, which serves as the mechanism of removal from the water. However, as part of the case-by-case authorization for the use of such agents, the Agency believes that consideration should be given to whether the entire agent or product would be consumed, or if combustion by-products in the air would settle or directly dissolve during the burn into the water column. The Agency requests comment on whether it should add ignition devices to the definition of a burning agent and if it should finalize a generic listing on the Schedule for these agents.

[2] Fingas, Mervin, (Ed) (2011), Oil Spill Science and Technology, Gulf Professional Publishing, pp. 836, ISBN: 978-1-85617-943-0.

Chemical agents - The Agency is proposing to revise the definition to read: "Chemical agents are elements, compounds, or mixtures designed to facilitate the removal of oil from a contaminated environment and mitigate any deleterious effects. Chemical agent categories include, but are not limited to, burning agents, dispersants, herding agents, sinking agents, surface washing agents, and solidifiers." The proposed language changes reflect the distinction the Agency is now making between chemical and biological agents, which will serve to better target the testing requirements for the product manufacturers and to better inform the OSCs in authorizing the use of agents in specific situations. The proposal also removes from the definition agent categories that are either being eliminated, prohibited or amended to conform with those changes, as discussed below.

Dispersants - The Agency is proposing to revise the definition to read: "Dispersants are typically mixtures of solvents, surfactants, and additives that promote the formation of small droplets of oil in the water column by reducing the oil-water interfacial tension." The proposed definition specifically addresses the mechanism of action through which these agents assist in mitigating the consequences of a discharge, clarifying for manufacturers which testing requirements they will be subject to when seeking to list a product on the Schedule.

Sinking agents - The Agency is proposing to revise the definition to read: "Sinking agents are substances deliberately introduced into an oil discharge to submerge the oil to the bottom of a water body." The current definition is ambiguous in regards to how much sinking below the water surface would qualify as a sinking agent, as some degree of sinking can also be associated with other agents, such as dispersants. The proposed definition specifies these agents sink the oil to the bottom of a water body. The Agency is prohibiting the use of sinking agents in the remediation of oil discharges in water.

Sorbents - The Agency is proposing to revise the definition to read: "Sorbents are inert,

insoluble substances that readily absorb and/or adsorb oil or hazardous substances, and which are collected and removed from the environment. Sorbents may be used in their natural bulk form, or as manufactured products in particulate form, sheets, rolls, pillows, or booms. Sorbents consist of: (1) natural organic substances (i.e., peat moss, straw, cellulose fibers, cork, corn cobs, feathers); (2) inorganic/mineral compounds (i.e., volcanic ash, perlite, vermiculite, zeolite, clay); and (3) synthetic compounds (i.e., polypropylene, polyethylene, polyurethane, polyester)." The proposed changes simplify the definition by removing the definitions of absorption and a confirmation that are embedded in the current definition of sorbents. The Agency believes this is appropriate given these are generally recognized scientific terms and the proposal does not distinguish sorbents or in any way restrict their use based on whether they absorb or adsorb the oil. The definition also adds the "natural" qualifier to organic substances, indicating that organic substances that have been treated with other substances would not necessarily fall under this category of agents and should not immediately be considered a sorbent. It also expands on and simplifies the examples by removing the references to the type of birds that feathers could come from, and by adding clay to the examples for inorganic/mineral compounds. The Agency requests comments on whether the definition is appropriate or if there are other materials that should also be included. Additionally, the Agency requests comments on particulate materials (e.g., clay) as sorbents, and whether their use should be specified only in booms or other manufactured products, but not as loose materials when authorized for use on water. The concern is for the potential for these materials to become sinking agents.

Finally, EPA also requests comments on the qualifier phrase "that are collected and removed from the environment." An organic and biodegradable sorbent (e.g., bagasse) may not necessarily need to be removed after it has absorbed/adsorbed the oil when used in, for example, a

wetland or salt marsh. Such removal may cause more harm than the oil itself due to trampling in the wetland or salt marsh. Once the oil is brought to the surface by the sorbent, biodegradation of both the oil and the sorbent can take place, especially if nutrients are added to enhance the biodegradation rates.

Surface washing agents - The Agency is proposing to revise the definition to read: "Surface washing agents are substances that remove oil from solid surfaces, such as beaches, rocks, or concrete, through a detergency mechanism that lifts and floats the oil for collection and removal from the environment with minimal dissolution, dispersion, or transfer of oil into the water column." The proposed changes clarify that these agents are to be removed along with the oil being treated from the environment.

b. New Definitions:

The Agency is proposing to add several new definitions that serve as the foundation for the new proposed biological agent classification: bioaccumulation, bioconcentration, biodegradation, and biological agents. They include basic terminology and address the action mechanism for biological agents and are consistent with definitions of these terms as generally understood by the scientific community. These new definitions are proposed to read as follow:

Bioaccumulation is the process of accumulation of chemicals in the tissue of organisms through any route, including respiration, ingestion, or direct contact with the ambient or contaminated medium.

Bioconcentration is the bioaccumulation of chemicals in the tissues of organisms from water alone.

Biodegradation is a process by which microorganisms metabolically decompose contaminants into biomass and simpler compounds such as carbon dioxide, water, and innocuous

end products.

Biological agents are microorganisms (typically bacteria, fungi, or algae) or biological catalysts, such as enzymes, able to enhance the biodegradation of a contaminated environment.

Bioremediation is the exploitation of microorganisms to convert contaminants into biomass and innocuous end products by the addition of materials into a contaminated environment to accelerate the natural biodegradation process.

The Agency is also proposing new definitions for the terms herding agents and solidifiers. The proposed definitions address types of agents originally captured under the surface collecting agent or the miscellaneous oil spill control agent categories respectively, and are terms that are more commonly used in industry. The definitions more specifically describe the action mechanisms for the categories and are consistent with definitions of these terms as generally understood by the scientific community. These new definitions are proposed to read as follow:

Herding agents are substances that form a surface film to control the thickness layer of oil and that are used to move oil horizontally across the water's surface.

Solidifiers are polymeric substances that cause the oil to become a coherent mass, preventing oil from leaching out or sinking into the water column, and which are collected and removed from the environment.

Finally, the Agency is proposing a new definition for product to clarify the difference between a specific product and an agent type or category. The new definition is proposed to read as follows:

Products are chemical or biological agents manufactured using a unique composition or formulation.

c. Removed Definitions:

The Agency is proposing to remove the definitions for surface collecting agent and for miscellaneous oil spill control agent (MOSCA). The surface collecting agent definition is being removed and replaced with a new herding agent definition to better reflect the common terminology used in industry for agents functioning through this type of mechanism. The MOSCA definition is being removed and replaced with a number of new and/or revised definitions for types of agents. The original MOSCA category was used as a catchall for all types of mechanisms of action. As the Agency adds new, more stringent testing requirements for listing products on the Schedule, there is a need for more specific category definitions to assist manufacturers in determining which of those testing requirements apply to their products. The Agency believes it has identified categories that capture all products to be listed on the Schedule; we request comment on whether the MOSCA category should be retained and whether the proposed categories are appropriate. If so, please provide sufficient information and rationale to support the addition of any new categories.

C. Subpart J - Use Of Dispersants, And Other Chemical And Biological Agents

1. General (300.900)

EPA is amending §300.900 by revising paragraphs (a) and (c). The proposed revisions clarify that Subpart J addresses not only chemical agents, but also those agents that now fall under the newly proposed biological agent category. The revisions expand on the notion that Subpart J is not only comprised of a Schedule of chemical and biological agents, but also includes testing requirements and authorization of use procedures. When taken together, these testing requirements, listing of agents, and authorization of use procedures address the types of waters and the quantities of listed agents that may be used in response to oil or hazardous materials discharges.

2. Authorization of Use (300.910)

The coordination of all applicable regulatory requirements is subject to ongoing efforts

among multiple agencies. While each regulatory organization is responsible for determining the scope of the preparedness requirements for vessels and facilities under their jurisdiction, preparedness activities for oil discharge situations are all interconnected through the Subpart J regulatory requirements addressing response methods. The Agency recognizes that the circumstances surrounding oil discharges and the factors influencing the choice of a response method or methods are many. While response actions are incident specific, the availability of response methods that adequately address the specific discharge situation depends largely on the contingency planning regulatory requirements and on how these requirements are implemented through the RCPs, ACPs and vessel and facility response plans. In order for a response to be effective, the NCP necessitates coordination between all these response plans, which can only be achieved through coordination between all the regulatory and planning entities. For example, USCG requires plan holders who store, handle, or transport oil and operate within areas or waters with plans that preauthorize dispersant use to identify and ensure dispersant response capabilities within those areas in their response plan (74 FR 45024, 45027). The Agency agrees with USCG's stated intent of this requirement to solely be ensuring dispersant capabilities, and the recognition that the regulatory requirements in Subpart J of the NCP govern the use of dispersants, or any other chemical or biological agent, as a response method for any discharge situation (74 FR 45012, 45015).

Although the NCP does not indicate a preference for chemical or biological agents over mechanical recovery devices, past Schedules have advocated the development and utilization of skimmers, booms and other mechanical control methods to remove or mitigate oil discharges from the environment with subsequent proper disposal (40 FR 6298). When developing contingency plans, the Agency continues to advocate response options that recover oil from the environment and

that result in the least environmental harm. The Agency has focused its regulatory efforts on chemical and biological agents considering their potential for short and long-term deleterious effects on the environment. Alternatively, because the Agency believes that the use of mechanical devices does not present a significant environmental danger in response to oil discharges, it has chosen not to regulate mechanical devices under Subpart J (59 FR 47407). Furthermore, the USCG has traditionally overseen the regulation of mechanical recovery devices and has worked together with the American Society for Testing and Materials (ASTM) to develop equipment standards for them. Additionally, DOI's Bureau of Safety and Environmental Enforcement has also developed equipment standards and facilitated research and development on mechanical devices. It is also in light of these efforts that the Agency continues to believe it unnecessary and duplicative to focus its regulatory efforts on mechanical recovery devices when other federal agencies are already addressing them (59 FR 47407).

Specific to dispersants, the Agency has emphasized that when making decisions on their use, OSCs, RRTs, and Area Committees should use the least harmful products that have been proven effective under the standardized laboratory conditions (59 FR 47411). More recently, due to the unprecedented use of dispersants in response to the prolonged continuous Deepwater Horizon discharge, the Agency policy for employing response countermeasures for that event prioritized the use of mechanical recovery over the use of subsea and surface dispersants use.[3]

EPA is amending §300.910 by revising paragraphs (a), (b), (c), (d), (e) and (f); and by adding new paragraphs (g), (h) and (i). EPA is not proposing major changes to the current authorities granted to OSCs, RRT representatives to the RRTs, States, DOC, DOI, or other National Response Team decision makers with regards to the authorization and application of chemical

[3] November 2, 2010, Mathy Stanislaus, EPA's Assistant Administrator to the Office of Solid Waste and Emergency Response (OSWER). Memorandum to Agency's Regional Administrators regarding revisions to ACPs and RCPs.

agents per current 300.910. However, EPA recognizes that the planning for and prolonged use and monitoring of chemical agents, especially dispersants, may require additional planning activities, discussion on authority to authorize, and monitoring requirements. Thus, the Agency is proposing to reorganize this section, adding titles to the paragraphs for ease of use, to add several requirements addressing the storage and use of agents, notification of agent use and recovery from the environment, and to revise language to clarify established EPA policy. The proposed revisions and clarifications are highlighted for each paragraph under §300.910 below.

EPA is also confirming, consistent with the intent of the NCP, that use of chemical or biological agents in response to oil discharges to waters of the US or its adjoining shorelines must be authorized by an OSC in accordance with Subpart J. The unauthorized use can result in violations of section 301 and 311 of the CWA. Section 301(a) makes unlawful "the discharge of any pollutant by any person," except in compliance with certain provisions of the CWA. In addition, 311(b) establishes penalties for persons who fail or refuse to comply with any regulation issued under section 311(j) of the CWA.

(a) *Use of Agents Identified on the Schedule on Oil Discharges Addressed by a Preauthorization Plan.*

The Agency is proposing revisions to paragraph 300.910(a) of Subpart J to address the preauthorized use of chemical and biological agents identified on the Schedule. The proposed revisions clarify the process for preauthorization, the responsibilities of all involved parties, and the factors to consider during the preauthorization process, including the authorization for the use of agents by the OSC at the time of a discharge. The Agency believes that the proposed revisions do not change its fundamental policies regarding roles of Federal, state and local representatives involved in planning for and responding to an oil discharge, but rather clarify the current

requirements and further explain the responsibilities for each party. The Agency is also proposing added procedures and review requirements intended to ensure preauthorization plans are up-to-date so they are most effective when implemented in case of a discharge.

EPA believes RRTs and Area Committees (ACs) must work together in order to effectively and successfully manage contingency planning. Thus, the proposed revised requirements continue to require that, as part of their planning activities, RRTs and ACs address the use of chemical and biological agents identified on the Schedule for discharge situations identified in preauthorization plans. Area Committees are responsible for developing ACPs, which must consider the use of dispersants or other mitigating devices and substances to prevent substantial threats and ensure the mitigation and effective removal of oil resulting from a discharge. ACPs must also describe the procedures for expedited decisions to use dispersants. RRT member agencies, including federal natural resource trustees, state agencies, USCG, and EPA, have delegated roles and responsibilities for oil discharges, and in some instances may fulfill the role of the Area Committees, facilitating both regional level and area-level contingency planning.

ACPs generally describe multiple discharge situations for the geographical area for which they apply. These situations are developed as part of area contingency planning activities, considering likely discharges from vessels, and onshore or offshore facilities, as applicable, and inform the development of preauthorization plans. EPA is proposing revisions to the term "specific context" in the current requirement for the preauthorization plan to clearly establish what "specific context in which products should and should not be used" refers to. The proposed language states pre-authorization plans must specify, for each identified discharge situation in which agents can be used, the maximum quantities to be used and the duration of use in any one event, the depth of water where it may be used, and the distance to shoreline where it may be used. The Agency

believes that clearly stating the use parameters that must be in a preauthorization plan will make it easier for planners to address concerns of preauthorizing agent use and in turn for responders to authorize their use.

In developing these parameters for the use of chemical or biological agents, preauthorization plans must at a minimum address the sources and types of oil that might be discharged, potential worst case discharges of oil, the existence and location of environmentally sensitive resources that might be impacted by discharged oil, inventory of the chemical or biological agents, storage locations and manufacturing capability of available agents, availability of equipment needed for agent use, availability of adequately trained operators, and the availability of appropriate means to monitor agent use. The Agency is revising the text only to simplify the language and clarify the requirements, but believes it is necessary to keep in place the fundamental elements that need to be considered.

The ACP must include procedures to identify and document the conditions under which dispersants or other chemical and biological agents can be used for the each discharge situations. Site-specific and oil-specific rationale for, and conditions/limitations to, the use of dispersants and other chemical or biological agents should be well documented as part of the plan. Information specific to the discharge situations identified, such as water mixing energy, water depth, direction and velocity of wind and currents, distance from shorelines, type of oil potentially discharged, potentially affected populations, and potential worst case scenarios for example, should be considered in establishing agent use parameters for preauthorization.

EPA is reiterating that RRTs and ACs must consider as part of the pre-authorization plans the potential short and long-term environmental effects on fish and wildlife, their habitat, and other sensitive environments that may result from the use of chemical and biological agents when

specifying their use parameters for each discharge situation. Information that can be considered includes, for example, data on:

- Benthic communities, marine mammals, sea turtles, coastal and marine birds, fish and shellfish, and plant life
- Threatened or endangered species and their critical habitat as defined by the Endangered Species Act of 1973;
- Sensitive biological resources or habitats such as essential fish habitat, refuges, preserves, special management areas identified in coastal management programs, sanctuaries, rookeries, and calving grounds; and
- Subsistence resources and harvest practices, recreational and commercial fishing (including typical fishing seasons, location, and type).

The NCP, for example, requires ACPs to include a Fish and Wildlife and Sensitive (FWS) Environments Plan identifying potential environmental effects on fish and wildlife, their habitat, and other sensitive environments resulting from removal actions or countermeasures, including the option of no removal. Information in the FWS plans may assist the RRTs and ACs when considering environmental effects in determining agent use parameters.

The Agency believes that preauthorization plans must consider not only the availability of agents, trained operators and equipment, but also the availability of appropriate means to monitor agent use at the time of a discharge situation. The Agency believes that it is important to monitor agent use to ensure its effective application and avoid any further detrimental effects its use may cause on the environment. Pre-authorization plans should consider a review process to evaluate the extended use of dispersants in a continuous discharge situation, including criteria for both

approving and suspending extended agent use. Monitoring the effectiveness and the impacts focusing on the local environment and/or the specific discharge situation would help inform responders of the appropriateness of the rates and amounts of dispersants used, and to adjust the operational conditions when necessary.

It is important to note that the NCP does not require pre-authorization plans to provide pre-approval for every discharge situation or allow unlimited use for discharge situations where pre-authorization is provided. Area Committees or RRTs may find that there is not sufficient credible scientific data to assess environmental tradeoffs and therefore not authorize the use of chemical or biological agents in certain discharge situations. Alternatively, they may authorize agent use in limited quantities and duration. The Agency believes that pre-authorization should only be provided for those discharge situations where there is sufficient scientific information that supports agent use, including that it will not cause more harm to the environment. Further, before authorizing the use of a chemical or biological agent, RRTs and ACs must consider the incident specific environmental trade-offs, as appropriate, with respect to the length of the response and size of the discharge. In analyzing environmental trade-offs, the RRTs and ACs must identify the affected biological resources and their habitats likely to be negatively impacted as well as those that are expected to benefit and document that the lack of availability of other response options that recovers oil from the environment.

While EPA is clarifying that RRTs and ACs may consider *in-situ* burns (ISB) in planning, burning agents cannot be pre-authorized for use. Section 311(d)(2)(G) of FWPCA and E.O. 12777 specifically provides that if dispersants, other chemicals, and other spill mitigating devices and substances are not identified on the Schedule that the Administrator may, on a case-by-case basis, identify any dispersants, other chemicals, and other spill mitigating devices and substances and the

waters and quantities in which they may be used. Burning agents are not listed on the Schedule and therefore authorization of use of burning agents is governed by paragraph 300.910(c) (See discussion on below on burning agents). Therefore, the Agency is proposing revising the language in this paragraph to remove the reference to burning agents, and to address them fully in paragraph 300.910(c).

As previously stated, all members of the RRT are afforded an opportunity to review and provide input to the AC on a draft preauthorization plan. However, only the RRT representatives from EPA and the state(s) with jurisdiction over the waters of the area to which the plan applies and the DOC and DOI natural resource trustees will have the authority to approve, disapprove, or approve with modification the draft preauthorization plan. The Agency believes this remains the correct approach. Given that preauthorization plans are developed during the contingency planning phase, the Agency believes that DOC and DOI natural resource trustee concurrence is preferred over just consultation because it provides for sufficient time to identify and resolve natural resource concerns. As noted in the 1994 NCP final rule, the requirement for concurrence during the advanced planning phase ensures trustee involvement in decision-making (59 FR 47398). Addressing in advance concerns that might otherwise slow the action ensures that operations during a removal action can be carried out quickly and effectively. The language regarding trustee concurrence in preauthorization plans for chemical or biological agents is also included in § 300.210(c)(4)(ii)(D) to meet the intent of section 1011 of the OPA, that there shall be consultation with "the affected trustees… on the appropriate removal action to be taken in connection with any discharge of oil." EPA believes natural resource trustee concurrence with pre-authorization plans satisfies the consultation obligation since the pre-authorization plans specify the use parameters for chemical or biological agents. Thus, the Agency is retaining this concurrence requirement for

preauthorization plans.

The Agency, however, considered adding specifically the EPA Administrator and the senior EPA representative to the NRT (e.g. NRT Chair) to paragraph 300.910(a) and other paragraphs to make clear the EPA Administrator's and senior EPA NRT representative's authority along with the OSC to direct any chemical or biological agent use. The Agency is proposing that if the preauthorization plan is approved in advance for chemical or biological agent use under specified discharge circumstances, then the OSC may authorize the use of the agents on the Schedule for their intended purpose without obtaining incident specific concurrences described in paragraphs (b) of this section unless otherwise directed by the Administrator as under current authority. While the Agency has previously proposed case-by-case authorization by the Administrator or designee until there was sufficient information to promulgate a Schedule of dispersants (47 FR 10978, 10995), the current requirement includes only the OSCs for authorizing the use of chemical or biological agents in discharge situations and for agents specified in the preauthorization plan. This was in response to comments that argued this approach enables rapid decision-making. However, the Agency considered whether it was necessary to clarify the existing authority delegated to the Administrator by Section 311(d)(2)(G) of FWPCA and E.O. 12777 by specifically including the EPA Administrator's and senior EPA NRT official's authority to direct chemical or biological agent use. The Agency believes this clarification would not impede rapid decision-making on the part of the OSC, and that for the majority of discharge situations, the OSC will remain as the sole authorizing entity. Note that in situations like a SONS or an event of extended duration, the Administrator already has the authority for, and is likely to have a more direct role in chemical or biological agent use decisions. Given these situations are rare and that this is an existing authority, the Agency is not proposing this clarification at this time. We request comment on this issue.

The Agency is making clear that chemical or biological agents may only be used for their intended use, given the different listing requirements proposed for the various categories of chemical or biological agents. For example, a chemical agent that is listed on the Schedule as a surface washing agent cannot be authorized for use as a dispersant, nor can a dispersant that is listed on the Schedule under saltwater conditions be used in freshwater. EPA is also including the term "incident" before "specific concurrence" to reflect that the preauthorization plan should already provide sufficient details to address the parameters for chemical or biological agent use for all identified discharge situations, so that specific concurrence for an incident described in the preauthorization plan is not required. If the parameters of chemical or biological agent use for a particular discharge situation are not fully described, or if the preauthorization plan is silent, then the discharge situation is not covered by the preauthorization plan and is accordingly covered under paragraph 300.910(b).

EPA proposes a new requirement for ACs to review and revise, if appropriate, preauthorization plans at least every 5 years to address revisions of the Schedule, new listings of threatened and/or endangered species, to reflect lessons learned from incidents and any other change that may impact the appropriateness of using chemical and biological agents that are listed on the Schedule. A 5-year review cycle is judged to be consistent with facility response planning requirements; as those plans are revised and updated, it seems reasonable that preauthorization plans should be reviewed and revised accordingly. The Agency recognizes that development of preauthorization plans can be resource intensive; however, once developed, a periodic review and revision as necessary should require much less effort. EPA welcomes comment on this timeframe and suggestions with supporting information for alternatives.

This review requirement is intended to ensure that preauthorization plans are actively

maintained and updated to reflect revisions to the Schedule. Preauthorization plans, as well as the facility and vessel response plans reflected in them, may include information on products listed on the Schedule. A review at least every 5 years is expected to provide greater consistency not only between any Schedule revisions, but also between any ACPs, facility and vessel response plans. For example, an ACP revision that results in a change in the worst-case discharge scenario would trigger a preauthorization plan review. Additionally, the requirement specifically includes plan review and revision requirements as appropriate to reflect new listings of threatened and/or endangered species that may occur. The Administrator or designated EPA RRT representative, the DOC and DOI natural resource trustees, and, as appropriate, the RRT representative from the state(s) with jurisdiction over the waters of the area to which a preauthorization plan applies must review and either approve, approve with modification, or disapprove any revisions to the preauthorization plans. This review and approval is intended to focus on any revisions, and is not intended as a requirement for review and approval for those portions that do not require modifications.

Finally, the Agency is proposing procedures for concurrence withdrawals. Any agency with concurrence authority that decides to withdraw concurrence from an approved preauthorization plan would be required to provide 30 days notice prior to withdrawal. The proposal calls for the withdrawal of concurrence to invalidate the entire preauthorization plan at the end of the 30-day notice period. Once the preauthorization plan becomes invalidated at the end of the 30-day notice period, the authorization of use for chemical or biological agents would need to be performed according to paragraph (b) of this section unless and until a new preauthorization plan is finalized under this paragraph. The proposal would require the RRT and the AC to address the withdrawal of approval of the preauthorization plan within 6 months of the withdrawal, allowing an opportunity to

address the concerns. The proposal also calls for the RRT and the AC to notify the NRT of the status of preauthorization in their area after the withdrawal of approval of a preauthorization plan at the end of this 6-month period. The Agency recognizes there may be other response planning authorities affected if a preauthorization plan is invalidated, which is in part why it is today proposing procedures to address this situation. Of note, USCG has responsibility for vessel and certain facility contingency planning. Included among USCG planning requirements for these facilities is a dispersant response capability that is contingent upon an approved preauthorization plan. In providing for these requirements USCG emphasizes that these regulations intend only to make dispersant equipment available (74 FR 45012). USCG also noted that "...if a particular area committee or regional response team is not satisfied that there is sufficient credible scientific data to assess environmental tradeoffs between dispersant use, shoreline cleanup, and mechanical recovery, then the committee or team is fully empowered not to allow the use of a dispersant response option, as authorized under 40 CFR 300, Subpart J" (74 FR 45012). Since pre-authorization plans must be approved by EPA, the DOC and DOI natural resource trustees, and, as appropriate, the RRT representative from the state(s) with jurisdiction over the waters of the area of the area to which a preauthorization plan applies, the Agency believes that a 30 day period of notice of intent to withdraw provides sufficient time for the USCG or any other affected plan-holder to make any modifications to their contingency planing activities. In the event of an Agency withdrawing its concurrence from an approved pre-authorization plan, EPA believes the advanced planning process should continue with consideration for all the elements specified in paragraph (a) of this section. While the absence of a preauthorization plan requires that authorizations for agent use be conducted according to paragraph (b) of this section, the Agency continues to believe that pre-authorization plans serve as a valuable advanced planning tool that provides a strong foundation to support

decision-making and strongly encourages the resolution of any withdrawal.

(b) *Use of Agents Identified on the Schedule on Oil Discharges Not Addressed by a Preauthorization Plan.*

The Agency is proposing revisions to paragraph 300.910(b) of Subpart J to address use of chemical or biological agents identified on the Schedule for discharge situations that have not been preauthorized. The proposed revisions clarify the authorities and responsibilities of all involved parties, and the factors to consider when authorizing the use of listed chemical or biological agents in these situations. The Agency believes the proposed revisions do not change its fundamental policies regarding roles of Federal, state and local representatives involved in an oil discharge response.

The proposed revisions maintain, with the appropriate concurrences and consultations, the OSC's authority to authorize the use of chemical or biological agents on the oil discharge, provided that the agents are listed on the NCP Product Schedule. The proposed revision to the concurrence authorization language would specifically include the EPA Administrator's concurrence authority to approve the use of chemical or biological agents on a discharge situation that is not addressed by a preauthorization plan. The concurrence of EPA representative to the RRT and, as appropriate, the concurrence of the RRT representatives from the states with jurisdiction over the navigable waters threatened by the release or discharge is maintained. The requirement for consultation with the DOC and DOI natural resource trustees, as appropriate, is also maintained. Additionally, the language is amended by replacing "when practicable" with "as appropriate" with respect to consultation with the DOC and DOI natural resource trustees. The Agency continues to believe that the case-by-case decision making process for OSCs must be flexible, minimizing the burden to the OSC of any consultations due to the time-critical nature of this process. However, advances in

communication technology (e.g., cell phones, email) provide OSCs with increased capabilities to communicate quickly. Therefore, the Agency believes it is reasonable to expect an OSC to have the capabilities to notify and explain the circumstances requiring use of the certain agents to natural resource trustees in a timely manner. Of note, while consultation with the trustees does not equate to a concurrence requirement, the Agency believes that such concurrence is highly desirable.

Similar to the proposed amended requirements under paragraph (a), the Agency is proposing to remove the term "specific context" in the current requirement for the preauthorization plan and instead clearly establish what the term refers to. The proposed language replaces the term with specific parameters that must be considered by the OSC for authorizing agent use. The proposed parameters include, but are not limited to, quantities to be used, duration of use, depth of water, distance to shoreline in which such agents are to be used, environmentally sensitive resources that might be impacted by discharged oil, agent inventory and storage locations, agent manufacturing capability, availability of equipment needed for agent use, availability of adequately trained operators and appropriate means to monitor agent use. These considerations are parallel to those proposed under paragraph (a) for preauthorization and planning purposes. The Agency believes these are fundamental elements that will provide key information on the overall ecological risks for the OSC to consider when authorizing the use of chemical or biological agents.

Finally, EPA encourages advanced planning for expedited decision making for cases where the discharge situation is not specifically addressed in the pre-authorization plans. Some RRTs have developed expedited approval guidelines that are not part of the pre-authorization plans, but that offer an opportunity for advanced contingency planning by gathering information on the key parameters discussed above. Because discharge situations not covered by pre-authorization plans need incident specific (i.e., case-by-case) authorization concurrence[4], expedited approval guidelines

can be used to support expedited incident specific authorizations. For chemical or biological agents listed on the Schedule that are not authorized for use under a pre-authorization plan, the ACP should outline the process established by the RRT for that region for expedited authorization decisions regarding their use. It is important to note that while the NCP requires that the ACPs include procedures for expedited decisions, these procedures can include disapproving the use of agents, or approving the use of agents with certain operational conditions. For example, areas may be designated in which the use of certain agents or other discharge mitigating devices is prohibited, situations where limits are placed on the quantities of agents used, or situations that require certain monitoring requirements are in place. Nonetheless, the Agency recommends that these expedited approval guidelines be developed, where appropriate, for use in situations where preauthorization is not in place.

(c) *Burning Agents*

As authorized under the FWPCA and delegated under E.O. 12777, the Administrator has the authority to, on a case-by-case basis, identify removal agents and other discharge mitigating devices not on the Schedule, and the waters and quantities in which they may be used. Consistent with previous Subpart J versions, today's proposed rule continues to provide for a case-by-case basis authorization of burning agents, rather than listing them on the Schedule. The proposal continues to allow for OSC's authorization of burning agents, with the required incident specific concurrences. The proposed revisions clarify the concurrence authority of the EPA RRT representative. The language maintains the concurrence requirements for the states, as appropriate. Additionally, the proposal clarifies that incident specific concurrences apply only to burns that involve the use of burning agents on water, and not to in-situ burning situations for which burning agents may not be

[4] http://www.epaosc.org/sites/5083/files/rrt6_nearshore_dispersant_eap_031605.pdf

necessary or for those not on water, such as land-burns.

In 2002, the USCG proposed burn credits to offset requirements for mechanical recovery, rather than establishing specific in-situ burning (ISB) response requirements. Requirements for ISB and associated burn credits were not included in the USCG 2009 final rule (74 FR 45006) to prevent the potential for reduction in the availability of mechanical recovery equipment. The USCG argued that allowing a credit for ISB might reduce the amount of mechanical recovery response equipment available in "areas where ISB pre-authorizations are in place." Further, that "ISB pre-authorizations in place provide sufficient incentive to encourage plan holders to stockpile ISB equipment if such equipment will be useful in addressing response situations…" The Agency is clarifying that it does not regulate the activity of in-situ burning for the purposes of Subpart J, but it does regulate the use of burning agents. However, burning agents may not be preauthorized for use under paragraph 300.910(a) of Subpart J. Consequently, OSCs need to approve the use of burning agents for in-situ burns with the required concurrences and consultations for each specific incident involving burning agents. Burning agents may only be authorized for use on a case-by case basis under paragraph 300.910(c). In-situ burns that do not use burning agents fall outside the Subpart J authorization of use requirements.

The Agency specifically addressed burning agents in the 1984 NCP final rule (49 FR 29195). While comments suggested that test data should be required for burning agents similar to that required for other chemical and biological agents, the Agency disagreed for several reasons. First, EPA argued burning agents are relatively common inorganic chemicals that generally exhibit low toxicity. Data on toxicity and other relevant parameters are included in standard references readily available to OSCs and requiring test data would be unnecessarily duplicative and burdensome. Second, state air pollution control programs generally regulate the use of burning

agents. Finally, EPA noted burning agents are used only rarely. However, several conditions have since changed. The Agency believes that burning agent technology has made available compounds beyond those that satisfy the current and proposed definitions of a burning agent, as discussed above. Further, the Agency recognizes that ISB has become an important response option that is used more frequently. For example, over 400 in-situ burns (resulting in an estimated 265,450 bbls of oil burned) were conducted during the Deepwater Horizon oil spill[5], and in-situ burns appear to be gaining a more prominent role as a response option in federal waters in remote locations, such as the Arctic. Therefore the Agency believes that it is important to maintain the requirement for burning agents to be considered by the OSC for authorization prior to use on a case-by-case basis, with EPA concurrence and the appropriate consultations with other federal and state entities. The Agency requests comments on this approach. The alternative of listing burning agents on the Schedule would require toxicity and efficacy listing thresholds for these products; the Agency is also requesting comments on test protocols that could be appropriately modified to serve this purpose.

(d) *Exceptions*

Today's proposed rule maintains the provision allowing the OSCs to authorize the use of any agent, including products not on the Schedule, when it is determined that the use of the agent is necessary to prevent or substantially reduce an immediate threat to human life. The proposed revisions do not change previous policy, but rather clarify the intent of the exception. The Agency believes that the protection of human life is the primary consideration in responding to an oil discharge. Life-threatening oil discharges (e.g., spills of highly flammable petroleum products in harbors or near inhabited areas) may occur at locations where chemical agents on the Schedule are

[5] http://www.restorethegulf.gov/release/2011/08/19/operations-and-ongoing-response-august-17-2011

not immediately available. The Agency believes that in such cases, an OSC must have the ability to use available agents that, in his professional judgment, would effectively and expeditiously mitigate the threat to human life. Allowing this authorization to occur without the required concurrences for preauthorization or authorization of use for products on the Schedule under paragraphs (a) and (b) respectively, or under paragraph (c) for burning agents, eliminates delays in responding to potentially life-threatening situations. While the OSC has this authority to use in response to immediate threats to human life, it must still inform the EPA RRT representative and the affected states of the use of an agent as soon as possible and must obtain their concurrence for the continued use of the agent once the threat to human life has subsided. This exception is not unbounded, and is intended only for those extraordinary situations in which time is of the essence to mitigate the threat to human life; revising the language to add the qualifier "immediate threat" emphasizes this authority is not intended to circumvent the authorization of use provisions in paragraphs (a), (b), and (c) of this section, which serve to address other human hazards situations.

The proposed revisions specify that concurrences must be sought for situations where continued use of chemical or biological agents extends beyond 48 hours. The Agency believes that advances in communication technologies (e.g., smartphones, cell phones, email) provide OSCs the increased capabilities to communicate quickly. Therefore, it is reasonable to expect that the OSC notify and explain the circumstances requiring use of the agent to the EPA Administrator or designated RRT representative and, as appropriate, the RRT representatives from the affected states and the DOC/DOI natural resources trustees within 48 hours. The Agency is requesting comments on these revisions.

(e) *Prohibited Agents.*

The Agency is maintaining the current prohibition for the authorization for use of sinking

agents, and is clarifying that this prohibition also applies to any other chemical or biological agent that, when mixed with oil, acts as a sinking agent. The Agency is also clarifying that the prohibition on sinking agents applies to those agents that sink oil all the way to the bottom of the body of water. While certain chemical and biological agents may partially sink oil from the water surface when applied to an oil discharge, these chemical and biological agents are not considered "sinking agents" for purposes of this prohibition if their mechanism of action does not completely sink oil to the bottom of the water body. Sinking agents, when applied to oil discharges, function by sinking floating oil to the ocean floor or the bottom of any other body of water where used, potentially causing adverse effects on benthic organisms vital to the food chain of the aquatic environment. Additionally, these agents are very hard, if not impossible, to remove.

The Agency is also adding a prohibition from listing on the Schedule and from authorizing for use any chemical or biological agents that have either nonylphenol (NP) or nonylphenol ethoxylates (NPEs) as components. This prohibition reflects the Agency's concern for these substances, as presented in the *EPA Nonylphenol and Nonylphenol Ethoxylates Action Plan,* released August 18, 2010. The Action Plan specifically addresses nonylphenol (NP) and nonylphenol ethoxylates (NPEs). NP and NPEs are produced in large volumes, with uses that currently lead to widespread release to the aquatic environment. NP is persistent in the aquatic environment, moderately bioaccumulative, and extremely toxic to aquatic organisms. NP has also been shown to exhibit estrogenic properties in in-vitro and in-vivo assays. NP's main use is in the manufacture of NPEs. NPEs are nonionic surfactants that are used in a wide variety of industrial applications and consumer products. Many of these, such as laundry detergents, are "down-the-drain" applications. Some others, such as dust-control agents and deicers, lead to direct release to the environment. NPEs, though less toxic and persistent than NP, are also highly toxic to aquatic

organisms, and, in the environment, degrade into NP. Both NP and NPEs have been found in environmental samples taken from freshwater, saltwater, groundwater, sediment, soil and aquatic biota. NP has also been detected in human breast milk, blood, and urine and is associated with reproductive and developmental effects in rodents. EPA is initiating both voluntary and regulatory actions to manage potential risks from NP and NPEs.[6] The Agency believes that this prohibition would not impose undue burden on product manufacturers to formulate their products, as there are safer viable alternatives to the use of NP and NPEs in product formulation.

The Agency also considered a broader prohibition for all endocrine disrupting compounds (EDC). Endocrine disrupting compounds are natural or synthetic substances, which through environmental or other developmental exposures alter the hormonal and homeostatic systems that enable organisms to communicate with and respond to their environment. Through numerous mechanisms of action, these substances directly impact the endocrine and reproductive systems. Effects of endocrine disrupting chemicals may be transmitted to further generations through genetic modifications or from continued environmental exposure of offspring. In recent years, laboratory studies have shown compelling evidence that endocrine systems of certain fish and wildlife have been affected by chemical contaminants, resulting in developmental and reproductive anomalies. There is strong evidence for adverse reproductive outcomes (infertility, cancers, and malformations) from exposure to endocrine disrupting substances, and mounting evidence for effects on other endocrine systems, including thyroid and metabolism. The group of molecules identified as potential endocrine disruptors is highly varied and includes chemicals used as industrial solvents or surfactants that can be found in dispersants and surface washing agents. Because of the common properties of these compounds and the similarities of the receptors and

[6] For more information refer to http://www.epa.gov/oppt/existingchemicals/pubs/actionplans/np-npe.html.

enzymes involved in the synthesis, release, and degradation of hormones, no endocrine system is immune to endocrine disrupting compounds. The Agency currently has no Subpart J testing or listing requirements for any product category on these effects. The Agency considered requiring literature reviews, modeling, and/or testing for those products containing potential endocrine disrupting compounds and those with the potential to produce EDC as degradation products. However, the lack of a definitive source that identifies potential compounds that have been tested and confirmed to have such effects is problematic.

The 1996 Food Quality Protection Act directed EPA to develop a screening program, using appropriate validated test systems and other scientifically relevant information, to determine whether certain substances may have hormonal effects in humans; the 1996 amendments to the Safe Drinking Water Act authorized EPA to screen substances that may be found in sources of drinking water for endocrine disruption potential. In response to these mandates, the Agency's Endocrine Disruptor Screening Program is developing requirements for the screening and testing of pesticides, commercial chemicals, and environmental contaminants for their potential to disrupt the endocrine system.[7] Although EPA has some data on endocrine-disrupting pesticides, insufficient scientific data are available for most of the chemicals produced today to allow for an evaluation of endocrine associated risks. The science related to measuring and demonstrating endocrine disruption is relatively new and validated testing methods are still being developed. When complete, EPA will use these validated methods or assays to identify and characterize the endocrine activity of pesticides, commercial chemicals, and environmental contaminants, specifically in relation to estrogen, androgen, and thyroid hormones. EPA plans to use the assays in a two-tiered screening and testing process: Tier 1 will serve to identify chemicals that have the potential to interact with

[7] http://www.epa.gov/endo/pubs/edspoverview/background.htm

the endocrine system; and Tier 2 will determine the endocrine-related effects caused by each chemical and obtain information about effects at various doses. With this two-tiered approach the Agency will gather information needed to identify endocrine-active substances and take appropriate action, as mandated by Congress. When all tests are finalized and chemical information begins to be collected, EPA may further consider how to incorporate that information into the Subpart J NCP requirements.

Because of the strong clear evidence for NP and the NPE as endocrine disruptors, the Agency is proposing a prohibition specific to products containing NP and NPE as components. However, the Agency is not proposing to include a broader prohibition on products that contain EDCs, but is requesting comments on expanding the prohibition to all potential EDCs, or to what the criteria for a broader prohibition should be in order to make that determination. As an alternative, the Agency could require all product components be tested for potential endocrine disruption and is requesting comment on this approach as well.

The Agency believes that the proposed prohibitions are applicable in all cases, notwithstanding the proposed provisions for case-by-case use authorization of burning agents under §300.910 (c), or for authorization of use of any chemical or biological agent when it is necessary to prevent or substantially reduce an immediate threat to human life under §300.910 (d). There are chemical and biological agent alternatives to sinking agents and to agents containing NP or NPEs, as well as mechanical methods for responding to oil discharges, including those situations that pose extreme threats or are time critical.

(f) *Storage and Use of Agents*

Current data requirements under Section 300.915 requires information be provided on recommended conditions of storage and use for each product at the time the application for listing a

product under Subpart J is submitted to the Agency. This information is summarized in EPA's *NCP Product Schedule Technical Notebook*. Specifically, this information includes: special handling and worker precautions for storage and field application; maximum and minimum storage temperatures (optimum ranges and temperatures that will cause phase separations, chemical changes, or other alterations to product effectiveness); shelf life of the product; recommended application rates and procedures, concentrations and conditions (considering water temperatures, salinity, types and ages of the oil); and any other application restrictions.

The Agency proposes regulatory revisions that complement the existing information requirements for the person or entity submitting a product for listing ("submitter"). The proposed requirements focus on the use of this information by the responder and the OSC. Specifically, the revised provisions require the OSC to only authorize for use those products that are stored under the conditions specified by the submitter, including the maximum, minimum and optimum temperatures, humidity and any other relevant conditions. Additionally, the Agency proposes to require that the OSC only authorize for use those products whose date of use does not exceed the expiration date listed on the container's label at the time of an incident, unless the responsible party provides the OSC documentation, developed in consultation with the submitter, prior to product use affirming the product has maintained its integrity, including no changes in the composition, formulation, storage conditions, effectiveness, and toxicity of any product. In such cases where the product expiration date has passed but the owner or operator of the facility/vessel storing the product still wants to be able to use the product inventory, EPA proposes that samples of the expired product lot representative of all storage conditions at any end user location be tested. Only if the owner or operator can demonstrate that the product has maintained its integrity is the OSC allowed to authorize the use of that product inventory. The owner or operator of the facility/vessel,

or their representative, must re-label the tested product lots and maintain test results documentation until those lots are used or discarded, and must retest the product at least once every 5 years to ensure effectiveness and to allow an OSC to authorize the product for use. The intent of these proposed revisions is to ensure that products being authorized and used have maintained their effectiveness, even through prolonged storage. The Agency believes it is the owner or operator's responsibility to ensure that any product it stockpiles for future use has maintained its effectiveness and has not degraded into more toxic byproducts once the manufacturer's expiration date has passed. This retesting provision is supported by the proposed requirements that the submitter provide not only the recommended storage conditions, the anticipated shelf life of a product at those conditions, and that the product label include both the manufacture and expiration dates, and conditions for storage. The flexibility in this proposed revision allows for the use of available product inventory that may still be viable.

(g) *Supplemental Testing and Information.*

In preamble to the 1994 amendments to the NCP (1994 Amendments), the Agency stated that when making decisions on the use of dispersants, OSCs, RRTs, and ACs should use the least harmful dispersants that have been proven to be effective under the standardized laboratory conditions. While at that time the Agency did not establish a toxicity threshold for dispersants, it is today proposing one for listing dispersants on the Schedule. The proposed threshold serves as a baseline, focusing the Schedule on dispersants that may offer a potential benefit in a response situation while minimizing harm to the environment. EPA continues to believe, however, that OSCs, RRTs, and ACs need to make decisions on dispersant use by weighing the toxicity data against other variables, including the effectiveness data for those dispersants. However, based on its oil spill response and contingency planning experience, the Agency has found that factors

impacting dispersant use decisions based on toxicity may vary relative to the target organisms and the environment in which the exposure occurs (e.g., variations of dispersant toxicological effects on species indigenous to a geographical area). Because of this, OSCs, RRTs, and Area Committees must be afforded flexibility when making dispersant use decisions based on these toxicity factors, and need to have toxicity data that will allow them to rank the various dispersants available based on acute toxicity (59 FR 47411).

The 1994 amendments to the NCP provided the RRTs the authority to require supplementary effectiveness and toxicity testing when developing preauthorization plans in order to obtain data more targeted and relevant to the area-specific and site-specific conditions of oil discharges for which they are responsible. The 1994 Amendments authorized the RRTs to require product manufacturers to conduct supplementary effectiveness or toxicity testing, or to provide available data or information beyond what is currently required in Appendix C, to address site- or area-specific concerns, such as sensitive indigenous species with critical roles in the local environment or that have special commercial value. This authority provides the opportunity to the RRTs and ACs to get more relevant information for their contingency planning efforts. The Agency continues to believe that additional testing should not be a mandatory requirement, as situations may exist where additional testing would not be necessary or would place an unnecessary regulatory burden on both the RRTs and the product manufacturers, vendors or, facility and vessel owners or operators.

Although EPA provided this supplemental testing option in the 1994 Amendments, the use of this option has been less comprehensive than envisioned at the time. Thus, today's proposal expands this authority by removing the limitation of requesting additional testing for preauthorization planning activities only. The revised language would allow the RRTs to require

additional testing on products in discharge situations outside preauthorization planning, including expedited decisions, or in discharge situations where there is a need to re-evaluate the extended use of a product and its effect on the area's ecosystem. EPA is also clarifying that any representative of an agency to the RRT with concurrence authority on the use of chemical or biological agents may request supplementary testing. For example, exploratory drilling in a new sale lease area may occur in a region that either may not be covered by a preauthorization plan (e.g., frontier areas) or the specific discharge situation may not be addressed in existing pre-authorization plans. Since EPA must concur before the OSC authorizes the use of a chemical or biological agent in that instance, the Agency may require additional information or supplementary testing that is more specific and relevant to the area or site conditions. Ideally, requests for supplementary testing or additional information would occur before a discharge occurs, but it may also occur during a response action. In requiring additional testing or information, the Agency may consider environmental impact analyses described in the exploration plan or other environmental assessments or impact statements, as necessary.

Today's proposal allows the RRT to request supplementary testing, available data or information from the owner or operator of a facility or vessel with a response plan identified under §300.211, clarifying its authority for such requests to the submitter. Supplementary toxicity and effectiveness testing, available data, or information may be requested to address specific site, area, or ecosystem concerns relative to the use of a product, including for example, acute and sub-chronic toxicity tests using native species and/or product effectiveness tests using site or area specific oils. Many factors influence the effectiveness of a chemical or biological agent, including for example oil composition, sea energy, state of oil weathering, type of dispersant used and the amount applied, temperature, and salinity of the water[8]. Owners or operators of affected vessels or facilities (e.g.,

offshore facilities) may be more familiar with specific site or area information regarding the types of oil and the affected environment. Having the authority to request supplementary information from an owner or operator would help RRTs develop plans for expedited decision making in discharge situations not covered by pre-authorization plans.

EPA considered extending the authority to request supplementary testing or information to OSCs responsible for developing ACPs. Area Committees must develop an ACP for its designated area under the direction of an OSC. Providing an OSC charged with directing the development of an ACP with the ability to require supplementary effectiveness and toxicity testing or information may help Area Committees develop their contingency plans, including in areas where pre-authorization plans may not have been developed. However, EPA decided not to authorize the OSC to request such supplementary testing or information because even though the individual OSC may not be a member of the RRT, the agency or Department of which the OSC is a member is represented on the RRT. Thus, OSCs can solicit that their RRT representative request supplementary effectiveness and toxicity testing or information when developing an ACP. Working through the RRT enhances RRT and AC coordination and provides consistency between their corresponding plans, which is beneficial because the RCPs typically cover more than one ACP. OSCs developing adjoining ACPs would also benefit because they could better integrate plans with other ACPs, vessel, offshore facility, and onshore facility response plans (40 CFR 300.210(c)(3)(v)). Further, Agency or Department RRT representatives provide one point of contact for product submitters, or affected owners or operators, to respond to requests for supplementary information or testing. Finally, by having the supplementary information authority, all RRT members will be informed when approving, approving with modification, or disapproving any pre-

[8] The oil composition is the most important factor, followed closely by sea energy and the amount of dispersant applied. Fingas, M. (Ed.), (2011). Oil Discharge Science and Technology. Gulf Professional Publishing. pp 452.

authorization plan, or when concurring on the use of chemical or biological agents, even if they are not members of a particular AC. Authorizing RRTs to request supplementary testing or information from either product submitters, or owners or operators of vessels or facilities, allows the RRT to compare products on the Schedule so the least harmful products that have been proven to be effective under the standardized laboratory conditions are considered for preauthorization or authorization of use.

(h) - *Recovery of Oil and Agents from the Environment*

The proposed Schedule includes certain agent categories under which products can be removed from the environment: solidifiers, sorbents and surface washing agents. For those categories, the Agency expects that agents would be recovered from the environment to minimize any potential adverse impact. Today's proposal adds a new requirement that charges the OSC with overseeing the response activity to recover these products from the environment. Recovery activities after the use of these agents would include containment of the agents in the water, collection of the agents mixed with oil or any residual agent, storage of the oil-agent waste prior to disposal, and disposal of that oil-agent waste. While it is appropriate to have the OSC oversee that these activities take place, it is ultimately the responsibility of the responsible party to conduct these activities and ensure that the agents are recovered from the environment. The Agency requests comments on this proposed requirement.

(i) - *Notification of Agent Use*

The proposal includes a new requirement for the OSC to provide the EPA representative to the RRT with information after the use of a chemical or biological agent within 30 days of completion of a removal action. The information required in this notification includes the quantity of agent used during the response, the duration of the discharge event, the locations where the agent

was used, the ambient conditions during the response, the methods used to recover the oil and agents from the environment when applicable, and any detected adverse environmental impacts resulting from the event. The proposal allows this information to be provided in the OSC report to the NRC as required under Section 300.165 of the NCP, if such a report has been requested. While other existing notification requirements serve to activate an immediate response to an event, the proposed requirement gathers information that will be useful in evaluating the response from an environmental impact point of view. It will also inform the review of preauthorization plans and provide a basis for any necessary changes to improve environmental protection. The Agency is requesting comments on this proposed new requirement.

3. Monitoring the Use of Certain Agents (§300.913)

The goal of establishing a Schedule under the NCP is to protect the environment to the greatest extent possible from possible harmful effects caused by spill mitigating products used in response to oil discharges. As stated in the OPA Conference Report (H.R. Rep. 101-653, 101st Cong., 2d Sess. at p. 147 (1990)), in preparing the Schedule, "the President should consider the long and short term effects on the environment of spill mitigating devices and substances, and select those which are least harmful to the environment." Today's proposal takes into account this potential harm to the environment by establishing a regulatory approach under Subpart J that not only requires test data and information to establish a Schedule of chemical and biological agents, but that also incorporates procedures for authorizing the use of those agents and includes monitoring requirements for certain discharge situations. Because each oil discharge represents a unique situation with distinct conditions, it is often difficult to make field predictions of the potential harm to specific environments from data gathered from laboratory testing. Conducting field monitoring to gather site-specific information is an important tool that informs operational

decisions on the effectiveness of dispersants as a response option.

There are several reasons why monitoring requirements are necessary to minimize the potential harm when using mitigation agents. First, laboratory test protocols are not representative of all conditions that may be found in the field. The requirements for listing chemical or biological agents on the Schedule are based on laboratory tests that, while reflecting certain field parameters (e.g., water salinity) that allow for comparison between different products under the same specified laboratory conditions, do not provide information on the possible varied conditions found in the field. The test methods proposed in Appendix C provide a basic set of test procedures intended to establish baseline data for comparison of products. The Agency has sponsored research to develop laboratory parameters that better reflect field conditions. For example, EPA has investigated the biodegradation rates of dispersed oil at different temperatures, and has developed a baffled flask test to test for the efficacy of dispersants, all with the intent to better approximate real field situations. While today's proposed requirements reflect these advances in laboratory testing, each discharge situation will continue to be unique. Further, it would be impracticable for a manufacturer to test every type of oil for which a product may be used for chemical effectiveness and for toxicity on every species found that may be found before a product is listed on the Schedule.

Second, the Agency recognizes the current regulation gives RRTs the authority to request supplementary testing or provide available data or information on the effectiveness and toxicity of an agent. This supplementary information would address the specific conditions at the areas and sites within their geographical area of responsibility. For example, the existence of a sensitive indigenous species that plays a critical role in the local sensitive environment or has special commercial value may be considered by an RRT in requesting additional testing or available information specific to those species. However, any testing or data gathering effort initiated under

this authority for the purposes of planning or preauthorization would still have the limitation of being laboratory based and not necessarily representative of varied real field conditions encountered during a discharge incident, even when targeted to a particular area or jurisdiction.

Finally, the Agency is unaware regulatory requirements or procedures that specifically require facility owners or operators to have monitoring capabilities to evaluate the environmental effects of dispersants and dispersed oil. For example, while BSEE regulations requires offshore facilities to submit oil spill response plans (OSRPs) for approval, the requirement does not specify that the data submitted in such a plan needs to be relevant to the operational conditions of use; rather these OSRP generally rely on Subpart J toxicity data, which is not site or area specific.

The goal of using dispersants is to break oil into small droplets and transfer it into the water column for a more effective transport and dilution below the toxicity threshold of the biota present. However, this transfer unavoidably increases the exposure risk to the biological resources in the affected area. Monitoring dispersant use activity has provided critical information that has informed the daily operational decisions needed during oil discharge events. A cooperatively designed monitoring program for in situ burning and dispersants, the Special Monitoring of Applied Response Technologies (SMART) is currently used for operational incident response.[9] SMART relies on small, highly mobile teams that collect real-time data during dispersant and in situ burning operations to address whether the dispersant and dispersed oil concentrations at sensitive locations are exceeding levels of concern, and whether the dispersants are effectively dispersing the oil. State and federal governments in charge of the incident response use SMART data for operational decision-making.

[9] The SMART program is a joint project of these agencies: U.S. Coast Guard, NOAA, U.S. Environmental Protection Agency, Centers for Disease Control and Prevention, and the Bureau of Safety and Environmental Enforcement.

In general, SMART guidelines are based on the roles and capabilities of available federal, state, and local teams. While data collected following the SMART dispersant protocol may be useful for evaluating the dilution and transport of the dispersed oil, it does not monitor the fate, effects, or impacts of dispersed oil. Additionally, the Deepwater Horizon discharge event has identified the subsurface application of dispersants as an alternative technique in oil discharge response. For subsurface application, given that it would more immediately transfer oil and dispersant into the water column, monitoring is especially important in order to quickly characterize not only the application effectiveness for operational decision-making, but also the ultimate fate and effects of the dispersed oil in the environment.

The Agency believes that monitoring is necessary to determine the operational effectiveness of dispersants, and should transcend from their initial application to include the ultimate fate and effects of the dispersant and dispersed oil in the water column in order to further inform decisions on the use of dispersants. Monitoring the actual effectiveness of dispersant use requires both the development and implementation of detailed plans to provide those Agencies who have the responsibility of authorizing the use of dispersants the information they need for decision-making, particularly for prolonged or continuous applications. Monitoring plans would need to include not only those parameters that provide information on the application (e.g., platform's spraying parameters) and on whether or not oil is being entrained into the water column, but also on information that addresses the transport and fate of the dispersed oil. Monitoring the initial dispersant application without monitoring the fate and transport of the dispersed oil leads to an incomplete understanding of the overall effectiveness of the dispersant operation, the duration of which could continue for months, as evidenced by the Deepwater Horizon oil discharge. Adverse effects on biological resources of the water column and of the benthic community from exposures

to dispersed oil and dispersant depends on the length of time and concentration of the exposure, which in turn is dependent on the fate and transport of the dispersant and dispersed oil. Because these exposures will unquestionably vary depending on the discharge situation the Agency believes it is important that in certain discharge situations a monitoring plan be in place and executed, particularly for those exposure scenarios where a continuous oil discharge is continuously dispersed over the same area, or for subsurface applications.

In proposing these monitoring requirements, the Agency also considered that ACPs are required to include a detailed annex containing a Fish and Wildlife and Sensitive Environments Plan. This annex must identify potential environmental effects on fish and wildlife, their habitat, and other sensitive environments resulting from oil removal actions or countermeasures, including the option of no oil removal. Specifically, the annex must provide monitoring plan(s) to evaluate the effectiveness of different countermeasures or removal actions in protecting the environment. However, there are no specific requirements to have the active monitoring capabilities required to implement these plans, which today's proposal would provide.

Under the NCP the owner or operator of the facility/vessel from which a discharge or substantial threat of discharge occurs has an obligation to carry out fully effective response measures. Monitoring the application, fate and transport, and environmental impacts of dispersant use will inform the effectiveness of this response measure. Thus, the Agency is proposing a new requirement for a dispersant monitoring plan to address the environmental impacts, the sampling methods, and the equipment necessary to implement the plan when dispersants are used. The proposal requires the owner or operator of the facility to develop, maintain, and have the capability to implement a monitoring plan in the event of any subsurface dispersant use to an oil discharge.

The Agency believes that the subsurface application of dispersants would occur primarily at

offshore facilities engaged in oil exploration, development or production. Applying dispersants below the water surface is a new approach to mitigating oil discharges in which the behavior of dispersed oil is less understood relative to their surface application. Therefore, the subsurface application of dispersant is a response technique critical to the implementation of a monitoring plan that informs decision-making and reduces further environmental impacts. Owners or operators of offshore facilities engaged in oil exploration, development or production are familiar with the specific site environmental conditions, as demonstrated by the environmental impact analysis required by BOEM regulations, and should be able to rapidly mobilize containment and other response resources to minimize environmental impact.

The Agency is proposing a parallel monitoring requirement for facility/vessel owners or operators for surface dispersant use in response to oil discharges of more than 100,000 U.S. gallons occurring within 24 hours or that are continuous for more than 96 hours. While applying dispersants to the water surface is not a new approach to mitigating oil discharges, the Agency believes it is appropriate to monitor discharge events where dispersants are used in greater quantities or for an extended period of time. The Agency chose 100,000 U.S. gallons as a threshold criterion because the NCP classifies a discharge of more than 100,000 U.S. gallons of oil to the coastal waters as major, and constrained the 100,000 U.S. gallons release to within 24 hours to capture those discharges that are most likely to have catastrophic environmental impacts. The Agency believes monitoring the extended use of surface dispersants for discharge situations longer than 96 hours is appropriate since those situations result in longer exposure of biological resources to dispersed oil. Further, many acute toxicity studies use 96 hour for the exposure duration, including the acute toxicity tests using *Menidia beryllina* described elsewhere in today's proposal. The Agency recognizes there are other toxicity tests with shorter exposure durations (e.g., 24, 48, or 72 hours)

but believes 96 hours is a reasonable threshold given is common use as the exposure duration in dispersant testing. The Agency welcomes comments on the proposed surface application monitoring thresholds. Alternative thresholds presented must include rationale in order for the Agency to consider them for final action.

(a) *Monitoring Dispersant Use in Subsurface Waters 300.913(a)*

In 2010 Bureau of Ocean Energy Management, Regulation and Enforcement (BOEMRE)[10] issued a Notice-to-Lessees (NTL 2010-N10) addressing the adequacy of spill response and well containment resources, including subsea dispersant injection equipment.[11] Subsea dispersant injection equipment is now being designed to inject dispersants directly into the oil near the source of the discharge.[12] This type of application is intended to minimize dispersant dilution into water before the dispersant has an opportunity to interact with the oil.[13] This closer to the source application approach would reduce potential adverse environmental consequences from the use of excessive quantities of dispersants.[14]

However, integrating a subsurface dispersant application capability for offshore oil well blowouts into the mix of response options (e.g., mechanical recovery) also presents new challenges. The subsurface application of dispersants to an oil discharge was not discussed under existing pre-authorization plans. Furthermore, no protocols for environmental monitoring the use of dispersant applied or discharged in subsurface waters to support operational decision-making existed prior to the Deepwater Horizon oil discharge in 2010. Environmental monitoring of subsurface applications

[10] On October 1, 2011, the Bureau of Ocean Energy Management, Regulation and Enforcement (BOEMRE), formerly the Minerals Management Service (MMS), was replaced by the Bureau of Ocean Energy Management (BOEM) and the Bureau of Safety and Environmental Enforcement (BSEE) as part of a major reorganization.
[11] http://www.bsee.gov/Regulations-and-Guidance/Notices-to-Lessees-and-Operators.aspx
[12] http://marinewellcontainment.com/expanded_system.php
[13] NRC, (2005), Oil Spill Dispersants: Efficacy and Effects, The National Academies Press, pp 138, http://www.nap.edu/catalog.php?record_id=11283
[14] Ibid

requires not only monitoring capabilities that go beyond ambient conditions, particularly for challenging water depths, but also access to knowledgeable personnel.

The use of dispersants is an environmental trade-off among different habitats and species. Mixing dispersants with discharged oil in subsurface waters raises ecological concerns over the short-and long-term impacts on the affected environment. The goal of subsurface dispersant application is to transfer the oil into the water column, preventing it from reaching the water surface where it may be transported to the shoreline and impact biological resources on the water surface (e.g., water fowl) or on the shoreline (e.g., nesting turtles). However, applying dispersant to an oil discharge does not result in the physical recovery of dispersed oil from the environment. Physical recovery of oil reduces the potential exposure of aquatic organisms and species. Instead, dispersing oil increases the potential exposure of water or benthic organisms to the dispersant-oil mixture.

Ideally, decisions to use dispersants are informed by ecological risk assessments that capture all ecologically relevant factors and are supported by scientific research of the effects of dispersant use on the affected environment. However, making decisions on dispersant use is challenging when a discharge situation presents significant uncertainties and unknowns.[15] Environmental benefits analyses addressing environmental trade-offs associated with dispersant application should account for biological resources that may be affected. However, certain scientific information that may inform these analyses may be unavailable or incomplete, such as information specific to the subsurface application of dispersants. The Deepwater Horizon incident has prompted the Agency to reassess existing policies to account for the possible subsurface application of dispersants.

Today's proposed rule establishes requirements for the owner or operator of a facility/vessel to develop and implement a monitoring plan for the subsurface application of dispersants. The

[15] Anastas, P.T., Sonich-Mullin, C., Fried, B., (2010) Designing Science in a Crisis: The Deepwater Horizon Oil Spill, Environ. Sci. Technol., Vol 44, pp 9250–9251, DOI: 10.1021/es103700x

purpose of the subsurface monitoring plan is to identify and assess the potential adverse effects from the subsurface application of dispersant and the resulting dispersed oil to a representative and diverse group of indigenous species. Some requirements for the subsurface monitoring plan are incident-specific while others may be satisfied ahead of any incident. The subsurface monitoring plan would be submitted to the OSC; the OSC and the EPA representative to the RRT would also approve the monitoring plan before any subsurface application of dispersants is authorized. The proposal requires the owner or operator of a facility to develop, maintain, and be able to implement a monitoring program for any subsurface dispersant use in response to oil discharges. Applying dispersants below the water surface is a new approach to mitigating oil discharges in which the behavior of dispersed oil is less understood relative to their surface application. Likewise, the Agency believes monitoring subsurface use of dispersants is critical to inform response actions and to reduce the potential for further environmental impacts.

(1) *Key Biological Resources*

The Agency is proposing that the monitoring plan include a description of the potential impact of dispersed oil and dispersants upon dispersant application on key biological resources, trophic levels, and their habitat, including transient or migratory species, and other sensitive environments, and that the monitoring program identify methods to minimize the effects of dispersed oil and dispersants on these resources. This information will be used to inform decisions on the use of dispersants when determining which response option (e.g. mechanical recovery, dispersant application, in-situ burn) is appropriate.

Key biological resources can be identified consulting several sources of environmental information prepared for the outer continental shelf oil and gas leasing, exploration and development process to address NEPA requirements. BOEM follows a four-stage process for this

purpose, for which Environmental Assessments (EAs) or Environmental Impact Statements (EISs) are prepared. The EISs for the specific lease sales provide greater detail regarding specific planning areas and address a variety of location specific factors (e.g., water depth, sea floor topography, distance from shore, ecological communities, and the presence of threatened and endangered species and cultural resources). For those offshore activities, the concerns are for the possible impacts of oil and gas development on marine mammals, birds, fish and shellfish, and the natural habitats that support these resources, among others. Operators of offshore facilities must submit plans to BOEM for review and approval prior to any drilling or production activity that require certain environmental information, including biological environment reports and environmental impact analyses (EIAs)[16]. The biological environment reports focus on threatened or endangered species, marine mammals protected under the Marine Mammal Protection Act (MMPA), marine sanctuaries, and critical habitats designated under the Endangered Species Act (ESA), among others. EIAs must describe those resources, conditions, and activities that could be affected by the proposed exploration or development and production activities, including:

- The benthic communities, marine mammals, sea turtles, coastal and marine birds, fish and shellfish, and plant life;
- Threatened or endangered species and their critical habitat; and
- Sensitive biological resources or habitats such as essential fish habitat, refuges, preserves, special management areas identified in coastal management programs, sanctuaries, rookeries, and calving grounds.

EISs may contain information on the expected impact of certain response options to oil discharges

[16] http://www.boemre.gov/eppd/compliance/nepa/index.htm

(e.g., dispersants), particularly if they describe impact scenarios for catastrophic oil discharge events. For example, the EIS for planning areas in the Gulf of Mexico contains information regarding the biological resources that may be impacted by the subsurface application of dispersants to a catastrophic oil discharge[17].

Other relevant sources of information may be available. Federal and state environmental protection agencies may have environmental assessments or other types of information on biological resources and their habitats including databases (e.g. NOAA-NMFS Southeast Area Monitoring and Assessment Program, EPA - National Coastal Assessment[18]). In addition, studies conducted by operators of offshore facilities, and universities or other research-oriented institutions may describe the potential impact of dispersed oil and subsurface dispersant on biological resources at different trophic levels, including their habitat, transient or migratory species, and other sensitive environments, and identify methods to minimize the effects of dispersed oil and dispersants on these resources. The Agency requests comments on the requirement for a description of the potential impact of dispersed oil and dispersants upon surface and subsurface dispersant application on key biological resources. Any alternatives presented must include rationale in order for the Agency to consider them for final action.

(2) *Resolvable Oil Constituents*

As part of the monitoring program, EPA is proposing that resolvable constituents and biomarker compounds in the source oil, including volatile petroleum hydrocarbons and biomarker compounds, be identified. This will be useful in distinguishing the discharged oil from other sources of oil. Source oil is the unadulterated oil that serves as the basis for chemically

[17] DOI, Gulf of Mexico OCS Oil and Gas Lease Sale: 2012 Central Planning Area Lease Sale 216/222 Final Supplemental Environmental Impact Statement, Appendix B.
[18] http://www.epa.gov/emap/nca/

fingerprinting the oil. Understanding the source oil's composition and distribution of the PAHs, biomarkers, alkane, and other oil constituents is important as it serves as a reference for comparing analytical results from water samples collected in the field during a response, and can be used to determine if the oil found in a water sample is associated with the discharge source. The composition of the source oil also serves as a reference to determine the degree of weathering for oil in water samples collected as part of the monitoring plan. In addition, the composition of the oil found in a water sample can be compared to the source oil composition to determine if a correlation exists. Samples of source oil from subsurface discharges can be obtained using a gas-tight isobaric sampler.[19] Alternatively, other available techniques may also be considered.

Biomarker Compounds. Crude petroleum oil is a complex mixture of tens of thousands of individual hydrocarbons and nonhydrocarbons, which impart certain properties to the oil.[20] Biomarkers in crude petroleum oils may retain original carbon skeleton, which makes them useful in fingerprinting oils since they offer a high degree of specificity among crude petroleum oils.[21] Due to the wide variety of geological conditions under which they formed, crude petroleum oils generally exhibit unique biomarker compositions. Some biomarkers (e.g., hopane) may be highly stable and degradation-resistant relative to other petroleum constituents. Certain biomarker ratios may be useful in determining not only the source of discharged oil, but also in monitoring the degradation and the weathering of oils.

Volatile Petroleum Hydrocarbons. Crude oil may include volatile petroleum hydrocarbons (e.g., methane).[22] Research suggests these constituents may exert a biological oxygen demand in

[19] Jeffrey S Seewald, J.S., Doherty, K.W., Hammar, T.R., Liberatore, S.P., (2002) A New Gas-Tight Isobaric Sampler for Hydrothermal Fluids, Deep-Sea Research I Vol. 49, pp 189–196
[20] Wang, Z., and Stout, S., Oil Spill Environmental Forensics: Fingerprinting and Source Identification, Academic Press, (2007), pg 13.
[21] Fingas, M. (Ed.), (2011) Oil Spill Science and Technology, Gulf Professional Publishing, pp 99.
[22] Brooks, J.M., Wiesenburg, D.A., Burke, Jr., R.A., and Kennicutt, M.C., (1981) Gaseous and Volatile Hydrocarbon Inputs from a Subsurface Oil Spill in the Gulf of Mexico, Env Sci Techn, Vol 15, pp 951-959.

the water column if they are present.[23] Having sufficient dissolved oxygen in the water column is critical to avoid hypoxia, which may be detrimental to organisms living in the water column.[24] Thus, knowing the composition of the volatile petroleum hydrocarbons is essential to understanding the exertion these constituents have on the overall potential oxygen demand of the water column. Since oxygen depletion in the water column is an important parameter that continuously informs decisions to apply or alter the subsurface dispersant applications, the Agency believes that these constituents must be included in any source oil analysis.

The Agency requests comment on the proposed requirements for the resolvable constituents, biomarker compounds, and volatile petroleum hydrocarbons in the source oil. Alternatives posed must include rationale in order for the Agency to consider them for final action.

(3) *Resolvable Constituents from Other Oil Sources for Subsurface Monitoring*

The Agency is proposing that the subsurface sampling plan include a survey of known potential extraneous oil sources from nearby oil infrastructure and natural seeps to identify all resolvable constituents, including biomarkers, from other potential sources to distinguish them from the source oil. Background sampling prior to the subsurface application of dispersants provides a baseline to evaluate changes in oil constituents in the water column or sediment that could be attributed to the source oil. In addition to oil discharges, other major sources of oil inputs may include produced waters, land-based discharges, and natural seepage. Natural seeps provide the largest petroleum input to the offshore Gulf of Mexico.[25] Distinguishing between potential oil

[23] Kessler, J.D., Valentine, D.L., Redmond, M.C., Du, M., Chan, E.W., Mendes, S.D., Quiroz, E.W., Villanueva, C.J., Shusta, S.S., Werra, L.M., Yvon-Lewis, S.A., Weber, T.C., (2011) A Persistent Oxygen Anomaly Reveals the Fate of Spilled Methane in the Deep Gulf of Mexico Science, Vol. 331, pp. 312-315

[24] http://www.ncddc.noaa.gov/hypoxia/

[25] OCS EIS/EA BOEM 2012-058, Gulf of Mexico OCS Oil and Gas Lease Sale: 2012, Central Planning Area Lease Sale 216/222 - Final Supplemental Environmental Impact Statement, Vol I, Chpt 1-4, pp 3-13. http://www.boem.gov/BOEM-Newsroom/Library/Publications/2012/2012-058_vol_1-pdf.aspx

sources assists the OSC and supporting response Agencies in making operational decisions regarding dispersant use.

The Agency considered including a similar requirement for surface monitoring plans. The Agency rejected this approach because it is impracticable to conduct a survey of known potential extraneous oil sources from nearby oil infrastructure and natural seeps for vessels that continuously traverse over water. Furthermore, oil discharges on the water surface that originate from offshore facilities would already have such a survey in their subsurface monitoring plan. The Agency requests comment on including the requirement for resolvable constituents from other oil sources for subsurface monitoring plans only. Alternatives posed must include rationale in order for the Agency to consider them for final action.

(4) *Dispersant Application*

The proposal requires the monitoring plan to identify the dispersant(s), rationale for dispersant choice(s), the expected efficacy and the results of any effectiveness tests, a description of the dispersant application method, monitoring of the dispersant application for its effectiveness on a daily basis, the daily oil discharge flow rate, dispersant-to-oil ratio, and for subsurface the daily discharge flow rate of any associated volatile petroleum hydrocarbons. Additionally, the requirement would include a description of the equipment to be used including expected hourly application rates, capacities, and total amount of dispersant needed.

Currently, packages submitting dispersants for listing on the Schedule must provide recommended application procedures, concentrations, and conditions for using their dispersants. Furthermore the Agency is proposing that these procedures address, as appropriate, variables such as weather, water salinity, water temperature, types and weathering states of oils or other pollutants. Many submitters provide a dispersant-to-oil ratio (DOR) as recommended application amounts. For

example, a DOR of 1:20 means one part dispersant to 20 parts oil. Therefore the type and amount of dispersant needed depends on the type and amount of oil to be treated. Thus, it is appropriate for the monitoring plans to include the DOR for the specific type of discharged oil and other types of product information that may be appropriate for determining the appropriate dispersants and application ratios for the discharge situation. Further, to reduce excessive use of dispersants, the subsurface monitoring plan must include the daily oil discharge flow rate provided by the owner/operator of a facility or vessel, including a separate rate for any associated volatile petroleum hydrocarbons when applicable. Based on the oil discharge rate and the DOR, the dispersant application rate then may be determined for the specific discharge situation.

Hypoxia is an important consideration for using or continuing to use dispersant. The requirement for a discharge rate for volatile petroleum hydrocarbons will provide, along with the individual composition of the volatile petroleum hydrocarbons, the potential oxygen load on the water column from these constituents. Therefore knowing the potential oxygen load on the water column, which could potentially be increased with the use of dispersant, is an essential monitoring element and will inform operational decisions on the use of dispersants.

The monitoring plan must also include a description of the equipment to be used, including expected hourly application rates and capacities. Dispersant equipment must be able to maintain the expected hourly dispersant application rate without significant deviation from the set rate. The application rate must be monitored hourly and any deviation of more than 10 percent (i.e., ±10 %) must be reported immediately (see discussion on reporting requirements). The Agency is requesting this information to ensure dispersant application rates remain relatively constant for continuous discharges. Changes in dispersant application rates outside of established parameters might result in dispersants being over or under used, altering dispersant effectiveness. Significant

fluctuations in dispersant application rates could also indicate equipment malfunctions, requiring a reassessment of the response techniques.

The Agency also proposes to include a requirement for monitoring the operational effectiveness of the dispersant application on a daily basis. The operational effectiveness assesses the encounter probability of the dispersant application. To be effective, dispersants must be incorporated into the oil matrix, or the dispersant is otherwise unnecessarily lost in the environment. Therefore monitoring for operational effectiveness is necessary to avoid the dispersant not mixing with the oil, resulting in excessive quantities of dispersant used.

Finally the Agency is not including a description of the discharge rate for the associated volatile petroleum hydrocarbons as part of the surface monitoring plan requirements because these compounds would readily evaporate, making it unlikely they would be incorporated into the water column in significant quantities that would result in hypoxia. EPA welcomes comment on the proposed requirements for dispersant application. Alternatives must include rationale in order for the Agency to consider them for final action.

(5) *Oil Distribution Analyses*

The Agency is proposing that the subsurface monitoring plan include a description and results of oil distribution analyses, including the volatile petroleum hydrocarbons, to inform the daily representative sampling locations for monitoring. The description would indicate the behavior of multiphase plumes for subsurface waters, taking into consideration the effects of current velocities, direction and stratification from the ambient density gradient. The description should consider the oil composition, including volatile petroleum hydrocarbons and all other relevant factors such as the velocity and direction of subsurface currents that may affect the transport of dispersed oil.[26] Subsurface currents may be determined using acoustic doppler current

profilers (ADCPs). Among other factors, horizontal subsurface currents, the ambient density structure, and the discharged oil's droplet size-dependent rise velocity govern the transport of oil through the water column. To the greatest extent possible, the description would consider evaluating the behavior of subsurface oil through trajectory modeling to provide operational awareness of dispersant use, which can guide future operational monitoring and sampling in the near and far fields. The analysis of the available trajectory modeling may be useful in identifying the areas where the movement of subsea dispersants or dispersed oil may pose a risk to fish, wildlife and other biological resources, and other sensitive environments. Results from trajectory modeling for surface dispersant application would be used to forecast dispersed oil movement and advise monitoring vessels on the daily sampling locations and depths. Trajectory modeling would consider the general flow of the water column immediately beneath the area of the dispersant application.

Trajectory models evaluating the behavior of oil discharges, including hypothetical discharges, are commonly used to evaluate potential environmental effects. For example, BSEE regulations require an appropriate trajectory analysis specific to the area in which an offshore facility is located as part of the information that must be included in OSRP by plan holders for a "worst case discharge scenario". This analysis identifies onshore and offshore areas that a discharge potentially could affect and must reflect the maximum distance from the facility that the discharged oil could move in a given time period. BOEM uses oil spill trajectory modeling to address the likelihood of spill occurrences, the transport and fate of any spilled oil, and the environmental impacts that might occur as a result of the spill. However, neither BSEE nor BOEM has a regulatory monitoring requirement for trajectory modeling to inform environmental sampling

[26] Socolofsky, S. A., Adams, E.E., and Sherwood, C.R. (2011), Formation Dynamics of Subsurface Hydrocarbon Intrusions Following the Deepwater Horizon Blowout, Geophys. Res. Lett., 38, L09602, doi:10.1029/2011GL047174.

efforts.

Results from the trajectory modeling are important and would be used to forecast dispersed oil movement and advise monitoring vessels on the daily sampling locations and depths. The modeling results will also assist decision makers on the potential impact to biological resources from exposure to dispersed oil. For example, trajectory modeling analysis may suggest that certain biological resources are at risk, which may lead to an operational decision to modify dispersant application procedures or criteria. The Agency requests comment on requiring trajectory modeling to be included in monitoring plans. Alternatives posed must include rationale in order for the Agency to consider them for final action.

(6) *Sampling Strategy*

The proposal includes a requirement for describing in the monitoring plan the air, water, and sediment samples to be collected. Sampling must be conducted using standard operating procedures addressing the frequencies, depths and locations, or as specified by the OSC or the EPA representative to the RRT. The Agency believes sample collection should follow established standardized protocols that are reliable and defensible. An accurate assessment of environmental data depends on the reliability, timeliness, and integrity of the data collected. If the data collection effort is not properly planned, then the data set available to the OSC and EPA may result in increased uncertainty.

The plan would address not only the standard operating procedures for individual sample collection, but also the overall sampling strategy to determine sample location and depths. The standard operating procedures would describe the appropriateness of the sampling method; the equipment needed for sample collection; a description of potential interferences, problems that may be encountered, and corrective actions that would be taken; sample collection procedure including

the preparation steps, representative sampling considerations, and sampling steps for each sampling device used; sample preservation, containers, handling, and storage; decontamination of sampling equipment; and record keeping and quality control procedures. The elements above are generally described in various guidance documents on standard operating procedures for environmental sampling.[27,28] If the responsible party plans to use land based laboratories, the monitoring plan would also address the prompt transport of samples to laboratories in order to minimize weather or other operational delays. Samples should be handled in a manner consistent with the procedures set forth in the Quality Assurance Project Plan (QAPP) developed by the responsible party.

Additionally, the plan would describe the techniques and methods used to determine the horizontal and vertical boundaries of the dispersed oil plume in the event of an oil discharge. Once this is established, the majority of the sampling collection can be focused on known or suspected locations and depths of dispersed oil. Based on the initial water sampling results to characterize the boundaries of the dispersed oil, the sampling design can then be extended to include a sampling grid. Although reference stations would be incorporated into the sampling design that are close to the oil discharge, they should not result in concentrated data collection in a small zone that limits the utility of the data. The sampling design would use a targeted and adaptive approach that takes into account oceanographic data, modeling, and existing observations of water and sediment anomalies (fluorescent signals as an indicator of oil and dissolved oxygen as an indicator of biodegradation) to determine the extent and behavior of dispersed oil in the subsurface water. The plan would provide for the coordinated approach to sample collection among all monitoring vessels. The OSC and EPA would retain the flexibility within the monitoring and sampling plan to collect data or direct data collection based on their discretion. Additionally, the sampling strategy should

[27] http://www.epa.gov/quality/qa_docs.html
[28] http://www.epa.gov/region8/r8risk/sampling.html

be tailored to reflect the techniques and methods used to characterize dispersed oil plumes resulting from either surface or subsurface application of dispersants. The sampling strategy proposed would use targeted and adaptive approaches to assess the dispersion of the oil discharged and guide decisions on dispersant application. The approaches would include techniques such as modeling, field observations and sampling verification, and laboratory testing. For surface monitoring plans, the approaches should also consider oil weathering. The Agency requests comment on the requirement for a sampling strategy to be included in monitoring plans. Alternatives posed must include rationale in order for the Agency to consider them for final action.

Response Air Monitoring. The Agency is proposing to include an air monitoring requirement in the dispersant plan. Air monitoring would involve the collection and analysis of air samples that are distinct from other worker exposure air monitoring. Volatile organic compounds (VOCs) are associated with many products,[29] including volatile petroleum hydrocarbons that are common constituents in many crude petroleum oils, which may raise concerns for workers in close vicinity of the discharge site. The hazard to workers can be reduced by monitoring and using protective gear, including respirators, as well as limiting exposure through limited work shifts, rotating workers out of high exposure areas, and pointing vessels into the wind.[30]

Evaporation also plays an important role for many types of oil that could be discharged on the surface, and contributes to the weathering process. If VOCs were present in an oil discharge on the water surface, they would likely evaporate into the atmosphere. The objective of dispersant-related air monitoring is to determine if the subsurface application of dispersants affects the amount of airborne VOCs that are associated with the discharged oil traversing through the water column.

[29] http://www.epa.gov/iaq/voc2.html
[30] DOI, Gulf of Mexico OCS Oil and Gas Lease Sale: 2012 Central Planning Area Lease Sale 216/222 Final Supplemental Environmental Impact Statement, Appendix B, pp B15.

Some investigators suggest that dispersants can be used to control VOC emissions from discharged oil. These investigators hypothesize that retaining the oil in the water column could prevent VOCs from evaporating, but recognized dispersant use for VOC emissions control needed to be confirmed with scientific controls.[31] In addition, for oil discharged from a well blowout, the investigators expressed uncertainty regarding VOC concentrations of concern once the oil passes through the water column and reaches the surface.[32] Other investigators suggest certain VOCs could essentially dissolve into the water column and likely remain at depth, while other higher molecular weight aromatics may likely differently distributed through the vertical water column and/or could reach the atmosphere.[33]

Other potential emission sources may contribute VOCs. Airborne VOCs may originate from multiple sources associated with either oil-related activities (e.g., from loading vessels that store and transport oil)[34,35], or response related activities such as in-situ burning.[36] Surface capture vessels that collect oil that includes volatile petroleum hydrocarbons from a subsea containment system may also contribute to VOCs emissions. To distinguish VOCs associated with the discharged oil, air sample collection near the other potential source of VOCs and downwind if applicable, would evaluate the composition and concentration of each potential emission. Likewise, corresponding water sampling would be collected at the same time and location as air

[31] Curd, H. (2011) The Use of Dispersant for the Control of Volatile Organic Compounds, IOSC, Paper No. 2011-359, http://www.iosc.org/papers_posters/IOSC-2011-359-file001.pdf
[32] Ibid
[33] Ryerson, T. B., et al. (2011), Atmospheric emissions from the Deepwater Horizon spill constrain air-water partitioning, hydrocarbon fate, and leak rate, *Geophys. Res. Lett.*, *38*, L07803, doi:10.1029/2011GL046726.
[34] NRC, (2003) Oil in the Sea III: Inputs, Fates, and Effects, pp. 76-77, http://books.nap.edu/catalog.php?record_id=10388
[35] http://www.sintef.no/home/MARINTEK/About-MARINTEK/Services/Services1/VOC-Emission-from-Crude-Oil-carriers/About-VOC-Emission-from-Crude-Oil-Carriers/
[36] Fingas, M.F., Lambert, P., Li, K., Wang, Z., Ackerman, F., Whiticar, S., Goldthorp, M., Schutz, S., Morganti, M., Turpin, R., Nadeau, R., Campagna, P., Hiltabrand, R., (2001) Studies of Emissions from Oil Fires, IOSC, pp. 539-544, http://www.iosc.org/papers_posters/02103.pdf

samples are taken directly above the surface water where the discharge occurred. Because ambient factors (e.g., wind direction and velocity) that affect VOC concentrations can vary over time, all sampling would occur at the same time. The monitoring plan would describe the procedures to document sample collection times and locations, response vessel locations and alternate sources of airborne VOCs that may contribute to VOC emissions. Air samples from each potential source would be analyzed to identify individual VOC constituents in order to distinguish those associated with any other potential source from those associated with the discharged oil.

Controlling VOC emissions with the subsurface application of dispersants partially depends on the operational and chemical effectiveness of the dispersant application. In turn, the operational and chemical effectiveness of the dispersant application is contingent on the dispersant application rate optimizing the dispersant-to-oil ratio. Dispersants applied at a low rate (or at a variable rate) may be ineffective in retaining VOC-laden oil in the water column. Therefore, the Agency is proposing the monitoring plan describe the procedures necessary to conduct a comparative analysis of dispersant application rates and the VOC concentrations observed on all surface vessels and facilities in the affected area. The Agency believes these requirements are appropriate if consideration is to be given to the use of dispersant to control VOC levels.

The Agency requests comment on the proposed requirement to include VOC monitoring for both surface and subsurface monitoring plans. Alternatives presented must include rationale in order for the Agency to consider them for final action.

Baseline Sediment Monitoring. The proposal requires the subsurface monitoring plan to include a description of the procedures to collect baseline sediment samples prior to an oil discharge in the immediate vicinity where an oil discharge may occur and in any direction of expected transport of dispersed oil. The Agency believes that analyzing sediment samples for oil

constituents and characterizing the potentially affected benthic communities prior to impact from an oil discharge will support operational decisions on dispersant use by characterizing the affected environment.

Analyzing potentially affected benthic communities, including the microbial populations and oil constituents in the sediments near the immediate vicinity of an oil discharge and in the direction where dispersed oil is expected to travel, provides a baseline to compare with results from sediments samples collected during the subsurface application of dispersants. This serves to evaluate the effects of the dispersant used on the potentially affected benthic community. Sediments support many types of benthic resources. For example, much of the Gulf of Mexico seabed is comprised of soft sediments, which provides residence to a wide variety of soft-bottom benthos (e.g., microbes, metazoans) including some that form the base of the food chain for several species.[37] Hard-bottom benthic communities exist both on the continental shelf and in deepwater. Deepwater hard-bottom benthic communities include chemosynthetic and deep coral communities.[38] In addition, BOEM protects certain areas in the Gulf of Mexico that serve as habitat for hard-bottom communities (e.g., Flower Garden Banks National Marine Sanctuary) from disruption from oil and gas activities.[39]

Subsurface dispersant application may adversely impact soft- and hard-bottom benthic communities since dispersed oil can either be transported to those areas or because of the close proximity of the oil discharge to those habitats.[40] Characterizing sediments that support benthic communities prior to an oil discharge allows monitoring to determine if any oil layers form that

[37] DOI, Gulf of Mexico OCS Oil and Gas Lease Sale: 2012 Central Planning Area Lease Sale 216/222 Final Supplemental Environmental Impact Statement, Appendix B, pp B8, http://www.boem.gov/Environmental-Stewardship/Environmental-Assessment/NEPA/nepaprocess.aspx
[38] Ibid, pp B9
[39] Ibid, pp B8
[40] Ibid, pp B20-B25

may be associated with the oil discharge. According to a 2012 BOEM final supplemental environmental impact statement (final SEIS) for a lease sale in the Central Planning Area (CPA) of the Gulf of Mexico, oil could reach the seafloor in several ways including:[41]

- the dispersed oil particles may flocculate, which results in suspended particles collecting into larger suspended particles whose density increases such that they are no longer remain suspended in the water and, thus, sink to the seafloor,
- when water currents carry a plume to contact the seafloor directly, or
- where the dispersed oil adheres to other particles and sinks to the seafloor, which could result in a wide distribution of oil.

Oil that settles on the seafloor could have short and long term negative impacts (e.g., lethal and/or sublethal). For example, dispersed oil (including dispersed-oil contaminated sediments) deposited on the seafloor could smother immobile benthic communities. Dispersed oil deposited on the seafloor may expose small benthic organisms to low level exposure of oil constituents that could produce sublethal effects or result in depressed oxygen levels (e.g., hypoxia) in the sediments because of microbial oxidation of the deposited oil. In today's proposed rule, EPA recognizes that some degree of sinking can be associated with agents (e.g., dispersants) in the discussion on the definition of sinking agents. While dispersants are not intended to sink oil to the bottom of a water body, the Agency is concerned that the subsurface application of dispersants could result in some dispersed oil being deposited on the seafloor that could result in both short and long-term adverse effects.

Constituents commonly found in oil already may be present in some sediments. A 2009 published report on the northern Gulf of Mexico continental slope habitats and benthic ecology

[41] Ibid, pp B24

indicated that low concentrations of PAHs were present in some sediments surveyed.[42] The report noted that sources of PAHs in offshore sediments could include natural petroleum seepage, offshore production platform operations, spills, sediment transport and atmospheric deposition. Thus, assessing sediment for constituents that may be found in oil, even if those constituents are not present or present in low concentrations, prior to an oil discharge establishes a baseline to compare results from sediment sampling conducted during a response. The Agency believes that sediment characterization should consider other potential sources of oil when determining oil constituents in baseline sediments. The Agency requests comment on whether it should require analyses for specific constituents that may be present in the sediments, such as nickel and vanadium, and the rationale for any alternative proposal.

Ideally, baseline sediment samples are collected and analyzed prior to an oil discharge impacting an area in order to minimize the uncertainty associated with any drilling activities or other potential sources of oil. However, the Agency believes baseline sediment samples collected and analyzed within a previous five-year period from start up operations of any oil-related activity and updated every five years thereafter is a reasonable time frame to establish baseline results, provided that no accidental discharge of oil has occurred during that period. Thus, the development of a sediment collection plan should consider the relevant site conditions to determine where to collect baseline data. If, for example, the local subsurface current profile suggests that the oil discharge could be transported in certain directions, then it would be appropriate to optimize the collection of baseline sediment data in those areas. In developing a baseline sediment sampling

[42] Rowe, G.T. and Kennicutt II, M.C., (Eds), Northern Gulf of Mexico Continental Slope Habitats and Benthic Ecology Study: Final report, DOI Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA., OCS Study MMS 2009-039, (2009), pp. 165 http://www.data.boem.gov/PI/PDFImages/ESPIS/4/4842.pdf

plan, the owner/operator may consider using trajectory models that incorporate relevant site conditions and factors such as oil type to determine the expected direction of both the discharged and dispersed oil.

The Agency considered also proposing a baseline sediment sampling requirement for surface monitoring plans. The Agency rejected this approach because it is impracticable to collect baseline sediment for vessels traversing over water. In the offshore waters, oil dispersed on the water surface is not expected to significantly affect the benthic communities. Furthermore, the other surface monitoring requirements proposed would identify potential concerns from the surface application of dispersants, which would account for the potential impact on sediments and the associated benthic community. The Agency welcomes comments on the proposed approach for baseline sediment sampling for surface and subsurface monitoring plans. Alternatives presented must include rationale in order for the Agency to consider them for final action.

Baseline Water Sampling. The Agency is proposing that the subsurface monitoring plan must include a description of the procedures to collect baseline water samples throughout the water column prior to an oil discharge in the immediate vicinity where an oil discharge may occur and in the direction where dispersed oil may be transported. The Agency believes that sampling and analyzing the baseline water samples for oil droplet size distribution, oil constituents, and conditions to support microbial communities in the water column, among the other criteria listed below, prior to impact from an oil discharge will support operational decisions on dispersant use by characterizing the potentially affected waters. The results from baseline water sampling provide a reference to compare to results from water sampling taken during a response. Baseline sampling can also provide background information for petroleum hydrocarbons that may enter water from other sources. For example, petroleum hydrocarbon can enter the Gulf of Mexico from various

sources such as natural seeps.[43]. Therefore the Agency believes any water column characterization prior to an oil discharge must consider other potential sources of oil (e.g., natural seeps) when determining the presence of constituents commonly found in oil in baseline water samples.

Ideally, baseline waters samples would be collected and analyzed immediately prior to the impact of an oil discharge to minimize uncertainty associated with any drilling activities or other potential sources of oil. However, the Agency believes baseline water samples collected and analyzed once in every five years is a reasonable time frame to establish baseline results, provided that no accidental discharge of oil has occurred during that period which may affect the results. Thus, the development of a water-sampling plan should consider the relevant site data to determine where to collect baseline water samples. If the local conditions (e.g., subsurface current profile) suggest that the oil discharge could be transported in certain directions, then it would be appropriate to optimize the collection of baseline water collection in those areas. In developing the baseline water sampling, the owner/operator may consider using trajectory models that incorporate relevant site conditions and other factors such as oil type to determine the expected direction of the discharged and dispersed oil. The proposal for baseline water sampling specifically includes:

- In-situ droplet size distribution analysis, including the number and volume mean diameters between droplet sizes ranging from 2.5 to 2000 μm, with the majority of data collected between the 2.5 and 100 μm sizes. Droplet size distribution is an important component to understanding the chemical and hydrodynamic effectiveness of dispersants, and can be obtained with devices such as the Laser In-Situ Scattering and Transmissometry. Small oil droplets (e.g., diameters in the tens of microns range) have smaller rise velocities and tend

[43] DOI, Gulf of Mexico OCS Oil and Gas Lease Sale: 2012 Central Planning Area Lease Sale 216/222 Final Supplemental Environmental Impact Statement, pp 3-13. http://www.boem.gov/Environmental-Stewardship/Environmental-Assessment/NEPA/nepaprocess.aspx

to remain suspended in the water column, where they can become widely dispersed by turbulent diffusion and subject to biodegradation. In contrast, large oil droplets (e.g., diameters in the hundreds of microns range) will tend to rise to the surface.[44] The number and volume mean diameters are common metrics for characterizing the central tendency of droplet-size distributions.[45] The number mean diameter (NMD) is an average of droplet diameters, whereas the volume mean diameter (VMD) is the diameter of a droplet with the average volume (i.e., the mean of the volume distribution). The effective application of dispersants is expected to generate a larger number of small droplets, reducing the average droplet size.[46] The trajectory of oil droplets rising from a well blowout in deep water depends on the droplet size.[47] Effective dispersant application to an oil discharge is expected to alter the droplet size distribution, and thus change the trajectory of rising oil subject to horizontal subsurface currents and stratification. However, hydrodynamic and other environmental factors also influence the oil droplet size distribution. Mixing energy dictates the breakup of large oil droplets into smaller droplets and the depth of submergence of the droplets. For example, wave conditions also significantly influence the dispersed oil droplet size distributions, with the plunging breaking waves generated under experimental conditions producing the smallest oil droplets. Additionally, studies indicate that the type of chemical dispersant used is an important factor in controlling the dispersed oil droplet size distributions. Further, studies have shown that the chemically dispersed oil may result in a

[44] Li, M. and Garrett, C., (1998) The Relationship Between Oil Droplet Size and Upper Ocean Turbulence, Mar. Poll. Bull., Vol. 36, pp. 961-970.
[45] NRC, (2005), Oil Spill Dispersants: Efficacy and Effects, The National Academies Press, pp 58-59, http://www.nap.edu/catalog.php?record_id=11283
[46] Li, Z., Lee, K., King, T., Boufadel, M.C., Venosa, A.D., (2008) Assessment of Chemical Dispersant Effectiveness in a Wave Tank under Regular Non-Breaking and Breaking Wave Conditions, Mar Pollut Bull, Vol. 56, pp. 903–912.
[47] Daling, P.S., Moldestad, M.O., Johansen, O., Lewis, A., and Rodal, J. (2003) Norwegian Testing of Emulsion Properties at Sea - The Importance of Oil Type and Release Conditions, Spill Sci Technol Bull, Vol. 8, pp.123–136.

droplet size distribution that is not observed without the use of dispersants (i.e., physically dispersed oil), both in terms of a smaller volumetric mean diameter and a bimodal distribution for droplet size diameters below 100 μm.[48] Thus, droplet size distribution analysis provides important data that is useful in optimizing operational guidelines, modeling transport and fate, and potentially evaluating biological effects of chemically dispersed oil.

- Fluorescence measurements. When exposed to ultraviolet (UV) light certain compounds adsorb energy that can elevate electrons into an excited higher energy level. The wavelength that excites electrons into a higher energy state is typically referred to as the excitation wavelength. Electrons unstable at the higher energy state return to a lower energy state, emitting energy at longer emission wavelengths resulting in fluorescence.[49] Many organic compounds fluoresce at specific excitation and emission wavelengths that allows for identifying many of the components of dissolved organic matter and other compounds (e.g., crude oil) in seawater.[50] For crude petroleum oils, the aromatic compounds in the oil fraction are used to determine the specific excitation and emission wavelengths. When subject to excitation at 245-280 nm, polycyclic aromatic hydrocarbons (PAH) fluoresce over wavelengths of 310 to greater than 400 nm, depending on the number of aromatic rings in the structure.[51] Oil-specific fluorometers are tailored to measure the aromatic hydrocarbons

[48] Li, Z., Lee, K., King, T., Boufadel, M.C., Venosa, A.D., (2009) Evaluating Chemical Dispersant Efficacy in an Experimental Wave Tank: 2 - Significant Factors Determining In Situ Oil Droplet Size Distribution, Env Eng Scien, Vol. 26, pp 1407-1418

[49] Henry, C.B., Roberts, P.O., Overton, E.B., A Primer on In Situ Fluorometry to Monitor Dispersed Oil, IOSC 1999;246

[50] Coble, P.G., (2007) Marine Optical Biogeochemistry: The Chemistry of Ocean Color, Chem. Rev., Vol. 107, pp. 402-418.

[51] Bugden, J.B.C., Yeung, C.W., Kepkay, P.E., Lee, K. (2008) Application of Ultraviolet Fluorometry and Excitation–Emission Matrix Spectroscopy (EEMS) to Fingerprint Oil and Chemically Dispersed Oil in Seawater, Mar Pollut Bull, Vol. 56, pp. 677–685.

fraction in oil. Although this measure does not include all oil constituents, fluorescence is a valuable screening tool in a response, providing a rapid indication of potential dispersed oil in the water column.

- In-situ and ex-situ dissolved oxygen (DO) measurements. Dissolved oxygen is an important parameter to monitor the subsurface application of dispersants that informs operational decisions related to the subsurface application of dispersants, particularly where dispersed oil plumes are expected. Dissolved and/or dispersed oil represents an available carbon source for microbial oxidation, which can lead to a decrease in the oxygen content in the water column. The biodegradation of oil, including volatile petroleum hydrocarbons consumes oxygen in the surrounding water column under aerobic conditions. Dissolved oxygen could approach or exceed levels of concern (i.e., hypoxia) and thus suffocate affected organisms. Dissolved oxygen measurements are commonly measured during oceanographic survey cruises with an in-situ oxygen instrument that provides immediate results. However, an instrument that passes though a dissolved and/or dispersed oil layer could lead to questionable measurements if temporarily fouling of the electrodes occurs. Therefore, relying solely on measurements from in-situ oxygen instruments may lead to an erroneous interpretation of oxygen data. Some investigators reported that during rosette survey operations, the oxygen microelectrode reported localized oxygen minimum layers in regions, which were coincident with the dissolved and/or dispersed plume depth, but were lower than the corresponding ex-situ oxygen measurements. Thus, the proposed requirements include ex-situ DO measurements to be conducted using Winkler titrations.
- Analysis on each water sample collected from the vertical profiling survey to determine the total petroleum hydrocarbons (TPHs), individual resolvable constituents, including volatile

petroleum hydrocarbons, mono and polycyclic aromatic hydrocarbons, including alkyl homologs, and all biomarker compounds. TPHs can be measured using gas chromatograph (GC) equipped with a flame ionization detector (FID). TPH analysis measures normal alkanes, can be obtained relatively quickly, and provides a good measurement for initial screening on board monitoring vessels. However, GC-FID analysis does not provide the PAH and biomarker (e.g., hopane) specificity that is important to quantitatively identify oil constituents that may be present in the oil discharge.[52] PAHs and biomarkers are typically measured using GC/MS. Identifying the concentrations of PAHs and biomarkers assists responders in making informed operational decisions regarding the subsurface application of dispersants.

- Fluorescence intensity analysis on each water sample collected from the vertical profiling survey to determine the fluorescence intensity ratio (FIR). EPA proposes to employ FIR as a relatively simple and rapid means to assess chemically dispersed oil. Measuring the reduction in the intensity ratio after applying dispersant to the oil in seawater is an indirect estimate of the effect of chemical dispersion.[53,54] FIR is derived from the three dimensional excitation/emission matrix spectra (EEMS) acquired by combining multiple emission spectra obtained from oil exposed to individual excitation wavelengths. If the optimal excitation and emission bands are known, then the FIR may be derived from measuring the fluorescence intensity at two different fixed emission wavelengths (e.g., 340 and 445 nm) at

[52] Wang, Z., and Stout, S., Oil Spill Environmental Forensics: Fingerprinting and Source Identification, Academic Press, (2007), pg 268.
[53] Bugden, J.B.C., Yeung, C.W., Kepkay, P.E., Lee, K. (2008) Application of Ultraviolet Fluorometry and Excitation–Emission Matrix Spectroscopy (EEMS) to Fingerprint Oil and Chemically Dispersed Oil in Seawater, Mar Pollut Bull, Vol. 56, pp. 677–685.
[54] Kepkay, P.E., Yeung, C.W., Bugden, J.C.B., Li, Z., and Lee, K. (2008) Ultraviolet Fluorescence Spectroscopy (UVFS): A New Means of Determining the Effect of Chemical Dispersants on Oil Spills, IOSC, pp 639 – 644. http://www.iosc.org/papers_posters/2008%20107.pdf

a single fixed excitation wavelength (e.g., 280 nm). The FIR is calculated by dividing the fluorescence intensity of the shorter band (e.g., 340 nm) by the fluorescence intensity of the longer band (e.g., 445 nm). Studies suggest that adding dispersants to crude petroleum oil increases the fluorescence intensity of the longer emission band (e.g., 445 nm), which in turn drives down the value of the ratio (e.g., less than two). Although individual petroleum oils may have common spectral characteristics, the ideal three-dimensional EEMS for the discharged oil should be obtained from scanning the source oil. Therefore, the subsurface monitoring plan must include procedures to obtain the EEMS using the source oil to determine the optimal excitation-emission wavelengths.

- Analysis for biologically available nitrogen (e.g., nitrates and TKN) and total, purgeable, and dissolved organic carbon. Biologically available nitrogen is a common nutrient that is necessary to support oil-degrading microbial activity. However, the nutrient levels in receiving waters in which oil may be dispersed can vary. For example, Northern Gulf of Mexico waters on the continental shelf and slope are heavily influenced by the Mississippi and Atchafalaya Rivers, which are the primary sources of nutrients to the Gulf. Deep waters may have varying nutrient levels, depending on localized currents and associated eddies, which can vary seasonally.[55] Waters with sufficient nutrients are needed to support biodegradation of those oils that are amenable to microbial oxidation. However, waters that may contain sufficient nutrients to support oil-degrading microbial activity but don't have a sufficient oxygen budget may present an area of concern to monitor. Dispersed oil that is transported into those areas of concern may exert a potential oxygen load on the water column, which could require amending procedures or criteria for subsurface application of

[55] DOI, Gulf of Mexico OCS Oil and Gas Lease Sale: 2012 Central Planning Area Lease Sale 216/222 Final Supplemental Environmental Impact Statement, Appendix B, pp B16,

dispersant use. Total Organic Carbon (TOC) represents another carbon source that is available for microbial oxidation. Other sources of organic carbon may compete with oil as a substrate for microorganisms and represents an additional carbon source that could result in oxygen drawdown. Furthermore, when compared to other GC data, the purgeable and dissolved organic carbon would provide insight into the type and form of the organic carbon.

- Heavy metals analysis. Crude petroleum oil may contain heavy metals, including nickel and vanadium.[56] Dispersing oil may increase the bioavailability of certain associated heavy metals to marine organisms and therefore, it is important to have this information.
- For subsurface monitoring plans, an analysis for turbidity is a general measure of water clarity. Turbidity is determined by measuring how much material suspended in water decreases the passage of light through the water.[57] Suspended materials include soil particles (clay, silt, and sand), algae, plankton, microbes, and other substances. As mentioned earlier, dispersed oil interacts with particles in water where the dispersed oil adheres to other particles and sinks to the seafloor, which could result in a wide distribution of oil. This measure provides a quick assessment of suspended materials in receiving waters from other water bodies, such as rivers.
- For subsurface monitoring plans, an analysis for pH for each water sample taken. The pH is a simple standard measurement taken when analyzing water quality. Water samples with higher or lower pH may indicate that the sample is contaminated, or may indicate an unidentified contaminant.

[56] All, M.F., Bukharl, A., and Saleem, M., (1983) Trace Metals in Crude Oils from Saudi Arabia, Ind. Eng. Chem. Prod. Res. Dev. 1983, Vol 22, pp. 691-694.
[57] http://water.epa.gov/type/rsl/monitoring/vms55.cfm

- Water temperature analysis. Temperature is measured using an electronic thermometer on the CTD instrument. When combined with salinity and pressure, temperature measurements are used to calculate water density, which changes along the vertical profile. Along with the horizontal subsurface currents and the rise velocity of the dispersed oil, the ambient density gradient is an important factor in determining the behavior of dispersed oil in the water column as a result of the subsurface application of dispersants.
- Water column conductivity analysis. Conductivity is measured by passing a current through the water, conductivity-temperature-depth (CTD) instrument, which is then compared against known salinities to obtain the water salinity. When combined with temperature, and pressure, salinity measurements are used to calculate water density, which changes along the vertical profile. Along with the horizontal subsurface currents and the rise velocity of the dispersed oil, the ambient density gradient is an important factor in determining the behavior of dispersed oil in the water column as a result of the subsurface application of dispersants.

The Agency notes that the proposal allows for baseline water sampling plans to reflect the mobile operating conditions of vessels, and to identify suitable sites that are outside of the impacted area, but representative of the area where the surface application of dispersants occurs

Incident Specific Water and Sediment Monitoring The proposal requires the development of a water column and sediment sample collection plan to be initiated following dispersant use to collect samples in the immediate vicinity of the discharge, in any direction of likely transport as determined by relevant site data, and in any direction towards the shoreline(s) including at such depths and locations where dispersed oil may be present to provide a quantitative analysis of dispersant effectiveness and the timed distribution of dispersant and dispersed oil components from the discharge location, any location where oil surfaces, and any direction of any subsurface plume.

The proposal requires that all of the relevant elements specified for the baseline water and sediment sampling are included, as discussed above in this section. The Agency requests comment on the proposed approach to incident specific water and sediment monitoring. Alternatives must include rationale in order for the Agency to consider them for final action.

(7) *Sampling Equipment and Analytical Instruments*

The Agency proposes to require the type of sampling equipment and analytical instruments must be used for monitoring. The proposal also includes requirements for equipment calibration according to manufacturer's recommendations and recognized scientific practices, and for the equipment to be maintained in good working condition.

The conductivity-temperature-depth meter (CTD) is an electronic instrument package that is deployed over the side of a monitoring vessel and used primarily for sampling the physical properties of the water column. The instrument measures conductivity by passing a current through the water, which is then compared against known salinities to obtain the water salinity. The CTD also measures temperature with an electronic thermometer and depth with a pressure sensor. Salinity, temperature, and pressure are used to calculate water density.[58]

The CTD typically is attached to a large metal rosette wheel, which also includes a sufficient number of water-sampling bottles with individual volumetric capacities to collect water samples at various depths (i.e., vertical profiling) for all analytical analysis. Using water bottles with sufficient volumetric capacity is important in vertical profiling. Some types of water bottles (e.g., Go-Flo bottles) can avoid sample contamination at the surface by closing them when they are lowered into the water column and open below the water surface. As a result, these bottles are neither contaminated on deck of the vessel, nor as they are lowered into the water, by oil that may

[58] http://www.whoi.edu/instruments/viewInstrument.do?id=1003

be present on or near the water surface. The Agency believes these types of water sampling bottles are appropriate for collecting water samples, as opposed to other types of water bottles that remain open as they pass through the top of the water column. As the bottles descend through the water column their ends can be closed at the desired depths, including the depths where fluorometers indicate the presence of aromatic hydrocarbons. Water samples are retrieved from various depths for analytical chemistry measurements including but not limited to PAHs, biomakers, TPHs, VOCs, and oxygen content.

The proposal requires the equipment to provide real or near real time in-situ sampling capabilities to locate and analyze the subsurface dispersed oil, particle size, dissolved oxygen, methane and carbon dioxide in the water column. Fluorometers capable of detecting the discharged oil are required for measuring the dispersed oil[59]; equipment (e.g., Laser In-Situ Scattering and Transmissometry (LISST)) or a method that is capable of operating at depth is required to measure oil droplets at the water depths where the oil discharge occurs; and sensor(s) for dissolved oxygen, methane (sensitive to low concentrations), and carbon dioxide each are required that are all capable of real or near real time measurements.

The Agency is proposing that each monitoring vessel be equipped with a quantitative capability to monitor methane in the subsurface monitoring plan. Some studies suggests that methane and other lower molecular weight hydrocarbons may contribute to oxygen consumption through microbial respiration.[60,61] Thus, monitoring for the presence of methane is important when

[59] Conmy, R.N., R.F. Chen, C.E. Del Castillo, B. Downing (2012) Calibrations and Quality Control of In Situ Sensors. In: Fluorescence Applications in Aquatic Science. Eds: A. Baker and P.G. Coble, Cambridge University Press, In Press.
[60] Kessler, J.D., Valentine, D.L., Redmond, M.C., Du, M., Chan, E.W., Mendes, S.D., Quiroz, E.W., Villanueva, C.J., Shusta, S.S., Werra, L.M., Yvon-Lewis, S.A., Weber, T.C., (2011) A Persistent Oxygen Anomaly Reveals the Fate of Spilled Methane in the Deep Gulf of Mexico Science, Vol. 331, pp. 312-315.
[61] Valentine, D.L., Kessler, J.D., Redmond, M.C., Mendes, S.D., Heintz, M.B., Farwell C., Hu, L., Kinnaman, F.S., Yvon-Lewis, S.A., Du, M., Chan, E.W., Tigreros, F.G., and Villanueva C.J., (2010) Propane Respiration Jump-Starts Microbial Response to a Deep Oil Spill, Science, Vol. 330, pp. 208-211

considering factors that may contribute to oxygen drawdown. The water depth in which an oil discharge may occur is an important factor when considering the dissolution of lower molecular weight hydrocarbons into the water column from slow buoyant migration. Some studies evaluated the behavior of gas plumes, which may be present in the water.[62,63,64] In-situ instruments such as a membrane inlet mass spectrometer may useful analyzing low molecular weight volatile petroleum hydrocarbon such as methane.[65] In addition, there may be commercially available methods or instruments to detect methane in the water column, such as those used to detect methane leaks from underwater pipelines. The Agency invites comment on innovative in-situ analytical techniques or methods that can be used to characterize dispersed oil plumes. Alternative instruments or methods posed for consideration must include a rationale.

The Agency is also proposing that each monitoring vessel must be equipped to conduct Winkler titrations on discrete water samples collected from vertical profiling casts. Winkler titrations offer accurate measurements of dissolved oxygen and may be used to calibrate oxygen electrodes. The Agency believes Winkler titrations may provide confirmatory analyses on oxygen measurements provided by commercially in-situ oxygen probes (e.g., SeaBird Electronic 43).

(8) *Ecotoxicity Testing*

The proposal requires the subsurface monitoring plan to include an ecotoxicity assessment to determine how effective a dispersant is in reducing the overall toxicity in water. EPA is considering two different approaches. The first approach uses biological assays in two-phases. A

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[62] Camilli, R., Reddy, C.M., Yoerger, D.R., Van Mooy, B.A.S., Jakuba, M.V., Kinsey, J.C., McIntyre, C.P., Sylva, S.P., Maloney, J.V., (2010) Tracking Hydrocarbon Plume Transport and Biodegradation at Deepwater Horizon, Science, Vol. 330, pp. 201-204.
[63] Chen, F., and Yapa, P.D., (2004) Modeling Gas Separation from a Bent Deepwater Oil and Gas Jet/Plume, Jour Mar Syst, Vol. 45, pp. 189– 203.
[64] Daling, P.S., Moldestad, M.O., Johansen, O., Lewis, A., Rodal, J., (2003) Norwegian Testing of Emulsion Properties at Sea—The Importance of Oil Type and Release Conditions, Spill Sci Techn Bull, Vol. 8, pp. 123–136.
[65] Camilli, R., and Duryea, A.N., (2009) Characterizing Spatial and Temporal Variability of Dissolved Gases in Aquatic Environments with in situ Mass Spectrometry, Environ. Sci. Technol., Vol. 43, pp 5014–5021

rapid acute toxicity screening would be followed by sublethal tests for those samples causing lethality levels above the controls. Biological assays are being considered because they provide a direct measure of toxicity on the collected water and sediment samples, including the entire dispersant-oil mixture. The monitoring plan would describe the rationale for selecting the toxicity testing protocols and would establish protective thresholds. The rationale should consider the relevance of the protocol to the ecological receptors in the surrounding waters and sediments (e.g., deepwater pelagic and benthic resources) and their function in the affected ecosystem (e.g., prey to higher order organisms).

Environmental impact analyses for the applicable exploration/development/production plans, statements for lease sale areas, and any other relevant assessments can help guide the development of an ecotoxicity assessment by providing relevant information on the affected environment. The ecotoxicity assessment must consider the relevant site conditions that may affect the implementation of testing protocols (e.g., discharge site location, ambient site-specific conditions, ambient water temperature), test organism availability, and availability of testing equipment and/or laboratories, and the procedures to mitigate impeding factors. All testing would be conducted using standardized test protocols, where available. Standardized toxicity testing protocols are widely accepted and provide well-established procedures that increase the likelihood of identifying laboratories that are capable of performing the specified tests. Any proposed alternative methods that employ non-standardized testing would need prior approval from the OSC and the EPA representative to the RRT. Ecotoxicity assessments inform operational decisions on dispersant use. The results from the ecotoxicity assessment, when combined with other relevant ecological factors such as potential impacts to shoreline, may influence a decision by the OSC and/or EPA representative to the RRT to discontinue the subsurface application of dispersants.

The Agency is also considering an alternative approach to biological assays. The alternative approach would be to monitor ecotoxicity in the water column concurrently with dispersed oil sampling for fluorometry, particle size, and water quality (e.g., dissolved oxygen). The TPH (total petroleum hydrocarbons) concentrations in water samples taken vertically throughout the water column would be compared to TPH-based ecotoxicity benchmarks (EBs). Water samples collected for comparison of aqueous TPH concentrations to EBs would be analyzed within 24 hours of collection and reported within 48 hours. Sampling for the alternative ecotoxicity assessment approach would also be performed in areas without dispersant application to distinguish ecotoxicity associated with physically dispersed oil from that of chemically dispersed oil.

EBs would be derived using a species sensitivity distribution (SSD) approach. An SSD is a probability distribution of the sensitivity of a group of species to a toxicant.[66] SSDs would be developed for representative oils (e.g., crude oils) using existing acute toxicity values for mortality or immobility (e.g., 48 and 96 hr LC_{50}s) where sufficient species diversity are available (e.g., toxicity data for 10 or more species). The EB would be computed from the fifth percentile of the SSD as the hazard concentration 5 percent (HC5). SSD HC5s are considered protective of 95% of species, have been used by EPA for developing ambient water quality criteria, and are generally accepted by the international community.[67] Chronic toxicity benchmarks may be derived applying safety factors to the acute toxicity EBs.

If the Agency decides to adopt the SSD approach, it would likely still provide for the OSC or the EPA representative to the RRT the flexibility to require spill-specific toxicity testing using

[66] Aldenberg, T. and Jaworska, J.S., (2002) Uncertainty of the Hazardous Concentration and Fraction Affected for Normal Species Sensitivity Distributions, Ecotox and Environ Safety, Vol. 46, pp. 1-18.

[67] Barron, M.G., and Wharton, S., (2005) Survey of Methodologies for Developing Media Screening Values for Ecological Risk Assessment, Integrated Environmental Assessment and Management, Vol. 1, pp. 320–332

biological assays to measure directly the toxicity of collected water or sediment samples, if deemed necessary for extended application of dispersants. While a more immediate approach, the SSD results are based on a correlation to TPH measures, and do not necessarily reflect all of the oil constituents or contaminants that may be present in the sample. The Agency requests comments on the proposed and alternative approaches to ecotoxicity assessment. Alternative approaches must include appropriate rationale in order for the Agency to consider them for final action.

(9) *Monitoring Vessels*

The Agency is proposing to require the plan include a description of the resources necessary to carry out the monitoring plan, including the number of monitoring vessels, communication capabilities and plans, and a daily vessel timetable for the OSC. The Agency believes the subsurface monitoring plan must record the resources necessary to implement the continuous monitoring of the surface dispersant application. Depending on the scope of the response, more than one monitoring vessel may be required for continuous monitoring. Monitoring vessels may need to return to port to transfer water and sediment samples to land based laboratories for further analysis, to exchange crews, and to conduct maintenance, among other needs. Furthermore, more than one monitoring vessel may be required at anytime to conduct coordinated monitoring. Two or more vessels working in coordination at the same time would provide a useful frame of reference for data collected in separate locations at similar times. Finally, today's proposed rule establishes requirements to provide a daily vessel timetable to the OSC and EPA representative to the RRT to ensure the orderly transition of monitoring activities and to allow any adjustments in monitoring procedures or equipment for future cruises.

The proposal would require monitoring vessels to provide access to EPA representatives or any other response personnel designated by the OSC for oversight of monitoring and sampling

activities, allowing for operational assessments when carrying out monitoring activities. To facilitate oversight and maintain communication with the OSC and other response personnel, each monitoring vessel must be equipped with fully functioning voice and data communication capabilities. A fully functioning communication capability will facilitate the transfer of field-collected data to onshore response personnel for further analyses of dispersant effectiveness. The subsurface monitoring plan must provide a description of the procedures to follow when EPA and/or designated OSC response personnel identify necessary modifications in data collection. To facilitate the scheduling of oversight activities, the monitoring plan must describe the timeline for cruise duration and establish procedures that provide at least 48 hour advanced notice of vessel departures to the OSC and the EPA representative to the RRT. The advanced notice will allow EPA and other federal response agencies time to arrange for personnel to accompany monitoring vessels as appropriate. The Agency requests comment on this approach. Alternative approaches must include rationale in order for the Agency to consider them for final action.

(10) *Notification to Discontinue Monitoring.*

The Agency proposes to require the implementation of monitoring programs until the OSC, with the concurrence of the EPA representative to the RRT, determines that monitoring is no longer necessary. The Agency believes monitoring should continue to evaluate the affected water column and/or sediments even after dispersant use is discontinued, until the OSC, with the concurrence of the EPA representative to the RRT, determines that monitoring is no longer necessary. Based on monitoring results, an operational decision may result in the periodic discontinuation of dispersant application. In such cases, monitoring should continue in order to evaluate the recovery of the affected environment and determine if dispersant use should be reinitiated. Furthermore, continued monitoring after dispersant application ceases provides continued operational awareness of the

affected environment that can inform operational decisions. For example, monitoring the fate and transport of dissolved and/or dispersed oil informs natural resource trustees on potential resources at risk, allowing the trustees to evaluate potential response options such as avoidance tactics. The Agency requests comment on this provision. Alternatives must include appropriate rationale in order for the Agency to consider them for final action.

(11) *Daily Reporting*

The Agency proposes to require a description of the daily reporting procedures and elements, including immediate reporting, to the OSC and the EPA representative to the RRT. The daily reporting procedures and elements would describe procedures to immediately report any deviation of greater than or equal to 10 percent of the average (i.e., arithmetic mean) hourly dispersant application rate for the total dispersant volume authorized for that 24 hour period. The monitoring plan would also describe the procedures to immediately report the reason for any deviation, such as equipment malfunction. The Agency believes that inconsistent dispersant application rates confound data interpretation of the sampling results due to variations in the DOR. However, the Agency is providing a margin for variation within 10 percent of the hourly dispersant application rate to account for equipment performance. The Agency believes that a margin for deviation within 10 percent provides an adequate margin for variation in dispersant injection equipment without being overly restrictive.

The Agency also proposes to include in the monitoring plans a description of the procedures to immediately report any biological resources, including any threatened or endangered species or their habitat, which may be at risk based on dispersed plume trajectory modeling and/or results from daily sampling. Results from daily sampling (e.g., droplet size distribution) provide input data to refine predictions of the likely subsurface dispersed oil direction using trajectory modeling. In

addition, results from daily sampling (e.g., TPH) may lead to decisions to alter dispersant application to minimize impacts on affected biological resources.

Today's proposal would also require procedures for daily reports on all monitoring activities, including the results of all sampling and data analyses, including the specific hourly injection rate for the previous 24-hour period, all collected data and analysis of that data within 24 hours of collection, including documented observations, photographs, and video. For analyses that require time beyond 24 hours, the sampling plan must include a schedule describing when data will be available, but may not exceed seven days. Daily reports would be required to include a description of the estimated daily subsea transport of dispersed and non-dispersed oil, including locations and times of surfacing oil and dispersants, using the best available trajectory modeling available. Rapid sample analysis affords the OSC and EPA representatives with relevant data from multiple measurements to support situational awareness of dispersant operations and adjust dispersant operation if necessary. Monitoring vessels with deployment beyond the time allotted for sample analysis and whose analysis requires onshore laboratory support may choose to deploy additional vessels to collect samples and transport them onshore. EPA proposes to allow these analyses be provided within 120 hours to allow for toxicity testing that may require up to 96 hours to perform and 24 hours for sample transport. The Agency believes that providing a 120-hour window for certain analysis that may require additional time, subject to approval by the OSC and the EPA representative to the RRT, provides an adequate opportunity to conduct all requested analysis in a timely manner without being overly restrictive.

4. Data and Information Requirements for Product Schedule Listing (§300.915)

The Agency proposes to revise the data and information requirements for listing products on the Schedule in Subpart J. The proposed amendments revise the efficacy and toxicity testing

protocols and listing criteria for all chemical and biological agents on the Schedule. Additionally, the Agency proposes revisions to the requirements for general product information, confidential business information claims, submission package contents, EPA review and listing procedures, requests for decision review, changes to products, transitioning products from the current Schedule to the new Schedule, mandatory product disclaimer, and removal of products from the Schedule.

a. General Product Information - §300.915(a)

The Agency is proposing to consolidate general submission requirements applicable to all types of agents that may be listed on the Schedule in paragraph (a) of section 300.915. Subsequent regulatory paragraphs would include requirements specific to each of the product category. Additionally, the Agency is proposing to require some additional information, as applicable, regarding products that are submitted for Schedule listing consideration. The proposed general information requirements for all products are as follow:

- Submitter: Name, physical address, email, telephone number, identity of submitter (i.e., manufacturer, vendor, importer, distributor, designated agent for the manufacturer), and documentation of such identity.
- General Product: All name(s), brand(s), and/or trademark(s) under which the product is to be sold; Material Safety Data Sheet (MSDS); sample product labels for all name(s), brand(s), and/or trademark(s); chemical or biological agent categories under which the product is submitted for listing, including information on the product's mechanism of action when it is used as intended.
- Supplier: Names, physical addresses, emails and telephone numbers of the primary distributors, vendors, importers, and/or designated agent acting on behalf of the manufacturer.

- Product Storage: maximum, minimum and optimum temperature, humidity and other relevant conditions for product storage; impact on product performance if the product is not stored within recommended limits; and anticipated shelf life at the recommended storage conditions;
- Product Use: Recommended procedures, including product concentrations, application ratios, and conditions for use, addressing variables such as weather, water salinity, water temperature, types and weathering states of oils or other pollutants, and product and oil containment, collection, recovery and disposal.
- Environmental Fate: Known measured data and supporting documentation, on the persistence, bioconcentration factor, bioaccumulation factor, and biodegradability of the product and/or all of its components.
- Physical/Chemical Properties and Formulation: Physical state; flash point; pour point; viscosity; specific gravity; pH; citation of standard methods used; identity of each chemical component, including the specific chemical name; corresponding Chemical Abstract Service (CAS) Registry Number; the maximum, minimum, and average weight percent of each product component; and the intended function of each component (e.g., solvent, surfactant).
- Biological Agents (microbiological cultures, enzymes, nutrients, fertilizers): All microorganisms and weight percent by current genus and species, including any reclassifications; all enzymes and their International Union of Biochemistry (I.U.B.) number(s); Enzyme Classification (EC) code numbers; the source of each enzyme, units, and specific activity; name(s) and volume(s) of the nutrients and fertilizers contained in the product; citation or description of the methodology used to determine product components;

certification, including data, methodology, and supporting documentation, indicating that the product does not contain, at levels that exceed the National Ambient Water Quality Criteria lowest density value: bacterial, fungal, or viral pathogens or opportunistic pathogens including, but not limited to, enteric bacteria such as Salmonella, fecal coliforms, Shigella, Coagulase positive Staphylococci, and Beta Hemolytic Streptococci or enterococci.

- National Water Quality Standard Contaminants: Certification, including data, methodology, and supporting documentation, indicating that the product does not contain, at levels above National Water Quality Standards lowest acute value for aquatic life: (a) metals reasonably expected to be in the product, including arsenic, cadmium, chromium, copper, lead, mercury, nickel, zinc; (b) cyanide; and (c) chlorinated hydrocarbons, pesticides, polychlorinated biphenyls (PCBs), Polynuclear aromatic hydrocarbons (PAHs).
- Testing Laboratory Information: Name, address, contact name, email address, phone number; national or international accreditations.
- Laboratory Testing Data and Reports: All applicable information, data and analysis specified in the testing protocols, including raw test data and replicates, notes and observations, calculated mean values and standard deviations, summary of stock solution preparation, source and preparation of test organisms, test conditions, and chain of custody forms.
- Production capacity: Estimated annual production volume; average and maximum volume per day; time needed to reach that maximum production rate (days).
- Performance Capabilities/Benefits: Recognition received from if applicable; national and

international product testing or use data, recognitions (e.g., EPA's Design for the Environment), and/or certifications, informing the performance capabilities or environmental benefits of the product.

The proposed revisions group together and simplify the general submission requirements applicable to all product types. EPA believes reorganizing the general requirements in a central location will clarify which requirements are applicable to all submissions, and which are specific to their product type by including them in separate sections. While the majority of the requirements listed above are currently required to be submitted under Subpart J, the Agency is proposing revisions to several of the existing general requirements and several new data and information requirements. The Agency believes the proposed revisions and added new requirements will better characterize the nature of the product and will assist EPA in product listing decisions. The information will also assist the RRTs in their area planning activities, and the OSCs in authorizing the appropriate use of chemical and biological agents. Details on the proposed additions and revisions are as follow:

(1) *Identification and documentation by the submitter of its status in relationship to the product as the manufacturer, vendor, importer, distributor, or other designated agent of the manufacturer.* This proposed requirement is intended to clearly establish the point of contact the Agency will hold responsible for the submission, and to avoid any conflicts or claims from unauthorized entities on products listed or submitted for consideration.

(2) *Chemical or biological agent categories under which the product is submitted for listing, including information on the product's mechanism of action when used as intended.* Currently, when a product meets the definitions of two or more product categories, a submitter must provide technical product data for each of those product categories. After review of the submitted

technical product data, EPA makes a determination on whether and under which category the "mixed product" should be listed on the Schedule. It has been Agency policy to list products under one category only to avoid confusion in the field during a response, and because the different product categories' mechanisms of action do not generally allow for much overlap of multiple uses for one product. However, EPA has received comments from responders, vendors, and manufacturers who believe that it is appropriate for certain products to be listed under more than one category. The Agency is proposing to revise the current limitations for mixed products. The proposal allows for products to be considered under multiple categories provided they meet all of the listing requirements for each. EPA is soliciting comments on the issue of multiple category product listing.

(3) *Copy of the Material Safety Data Sheet (MSDS) safety document required by the Occupational Safety and Health Administration (OSHA) under their Hazard Communication Standard (HAZCOM) for the product.* EPA recognizes that chemical and biological agents may contain substances that could cause harm to oil spill responders who, if unaware of the product's formula, may not wear the proper personal protective equipment. MSDSs describe the hazards that may be involved with the product and recommend safety measures that would minimize or avoid adverse consequences that may result from exposures. The Agency believes MSDS information will be useful to both OSCs and responders when authorizing and using the product respectively, and that adding this new requirement for a product MSDS is appropriate. The Agency requests comments on this new requirement, whether the submitter should provide any other additional information on potential adverse human health effects based on the product's formula and application methods not captured in the MSDS, and how to best communicate this information to product end users.

(4) *Sample product label for all names, brands, or trademarked versions of the product that include the manufacture and expiration dates of the product, as well as the conditions for storage.* The Agency would be allowing flexibility in complying with this requirement by specifying that the submitter does not need to affix new labels to comply with this section if existing labels already convey the required information. This proposed requirement is not intended in any way to supersede any other federal labeling requirement in place (e.g., OSHA's HAZCOM). The proposed requirement is intended to assist the OSC in ensuring that the product used to respond to an incident is still viable and effective, and the OSROS or any other responder that is storing the product to ensure that their stockpile is viable and available to be authorized for use.

(5) *Environmental fate information.* The Agency is proposing to request any known measured data and supporting documentation on the persistence, bioconcentration factor, bioaccumulation factor, and biodegradability of the product and/or all of its components. The Agency currently has no restriction on use of persistent bio-accumulative products. Existing testing protocols and modeling approaches are considered to establish thresholds for listing based on bio-concentration factors (BCF) or bioaccumulation factors (BAF). These factors offer ratios of the concentration of a particular chemical in a biological tissue per concentration of that chemical in water surrounding that tissue, or in the environment respectively. The Agency considered a tiered testing approach, where the submitter provides known BCF/BAF information for listing purposes, but may be required to provide testing based on application rates authorized for use. That is, the lower application rates would only require existing information or the use of existing EPA models, while higher rates would require additional testing of fish, bivalves, and earthworms, as needed. The Agency also considered testing requirements after product use, including the monitoring of biota for chemical bioaccumulation.

Similarly, there are no restrictions on the use of environmentally persistent products. The Agency considered requiring the submitter to use estimation techniques/models, such as the EPA model EPI Suite™, which estimates environmental fate properties (breakdown in water or air, etc.) that can indicate where a chemical will go in the environment and how long it will stay there. A tiered testing approach for larger quantity/duration spills similar to the proposed amendments for bio-concentration and bioaccumulation was also considered. Given that the Agency can estimate these factors, it is only requiring that available information or data be submitted on the product rather than specific product testing, as specific product testing for these factors can add significantly to the testing cost for each product.

EPA believes this information is necessary to inform the OSCs when authorizing these products for use, given the potential for their extended use in significant quantities. The Agency requests comment on whether testing for products' bio-concentration, bioaccumulation and biodegradation should be required for listing purposes, and if so, on listing thresholds for BCF and BAF.

(6) *Removal of the incorporation by reference of the standards used to measure a product's physical/chemical properties*. This is proposed to be replaced with a requirement for a citation of the standard methodology used to determine these values. EPA believes that citing the standard methodology used to determine the required values is sufficient in lieu of specifying commonly recognized standard methodologies. The Agency believes it is appropriate to make this change given the new requirements for accredited laboratories to conduct testing.

(7) *Chemical Abstract Service (CAS) Registry Numbers for each chemical component in a product*. Each CAS Registry Number (CAS Number) is a unique numeric identifier that designates only one substance and links to information about the specific chemical substance. CAS

Numbers serve as an international resource for chemical substance identifiers and are used by scientists, industry, and regulatory bodies because each number is unique and specific to only one substance regardless of how many other ways the substance can be described. This will assist the Agency in better characterizing a product's components and assessing its mechanism of action prior to listing on the Schedule.

(8) *Certification that bioremediation agents do not contain bacterial, fungal, or viral or opportunistic pathogens at levels that exceed the National Ambient Water Quality Criteria lowest density value.* While providing information on these product contaminants is currently required for bioremediation agents, there are no threshold levels for product listing; a positive result for any of the above pathogens may raise concern but would not prevent the product form being listed on the Schedule. The proposed threshold requirement for these contaminants is intended to provide information for listing decisions that ensure the use of bioremediation agents that will not result in exceeding established water safety levels. The Agency believes that this information is necessary to determine if a product is suitable for listing, particularly for bioremediation agents, which could potentially be used at recreational beaches. The Agency requests comments on whether it should establish listing threshold for products based on this criteria, and whether the levels selected for certification are appropriate for this application.

(9) *Certification that the product does not contain, at levels that exceed the National Water Quality Standards lowest acute value for aquatic life: (a) arsenic, cadmium, chromium, copper, lead, mercury, nickel, zinc, or any other heavy metal reasonably expected to be in the product; (b) cyanide; (c) chlorinated hydrocarbons; (d) pesticides; (e) polychlorinated biphenyls (PCBs); and (f) polynuclear aromatic hydrocarbons (PAHs).* Providing information (i.e., upper

limit/concentration, detailed analytical methods, sample preparation) on most of these contaminants is currently required for all products, with no established threshold levels for product listing. The Agency will continue to require information on the methodology and the data and supporting documentation used to determine the levels of these contaminants in a product. The Agency, however, will not specify what analytical testing method the submitter should use to make these determinations, as it currently does for chlorinated hydrocarbons. This will allow the submitter flexibility for testing their product. Additionally, the Agency is proposing to require data on several new contaminants: pesticides, PCBs, and PAHs. The Agency's concern with pesticides as contaminants is mostly due to their potential use on organic sorbents (e.g., peat moss, corn cobs, and cellulose fibers). The concern for PCBs is for their toxicity and classification as persistent organic pollutants, having toxic effects such as endocrine disruption. PAHs are potent atmospheric pollutants, of concern because some compounds have been identified as carcinogenic, mutagenic, and teratogenic. The proposed threshold requirements for all of these contaminants is intended to provide information for listing decisions that ensure the use of any product will not result in exceeding established water safety levels. The Agency requests comments on whether it should establish a listing threshold for products based on these criteria, and whether the levels selected for certification are the appropriate levels for this application. The Agency also requests comments on whether there are any other contaminants that should be included for evaluation of a product prior to listing.

(10) *Removal of the requirement for laboratories performing the efficacy and toxicity testing for products to have prior experience specific to the required methodology.* Currently laboratories performing testing for products to be submitted for listing consideration are required to document previous experience with the specific testing methodologies in Appendix C to part 300.

The Agency believes it is more appropriate to require that laboratories be nationally or internationally accredited. Accredited laboratories are expected to be capable of following a prescribed testing protocol and good general practices, providing assurance that the test results will be reliable. The Agency is proposing to remove the requirement for laboratories to have previous experience specific to the methodologies in Appendix C, and instead require that the laboratories hold accreditations from recognized national or international organizations. National and international accreditation organizations include, for example, the International Organization for Standardization (ISO), and the Laboratory Accreditation Bureau (recognized in the US through the National Cooperation for Laboratory Accreditation (NACLA) and the International Laboratory Accreditation Cooperation (ILAC)). The ISO 17025, Laboratory Competence standard, identifies high technical competence and management system requirements to guarantee laboratory test results and calibrations are consistently accurate. NACLA is a national voluntary organization of regulators, accreditation bodies and laboratories cooperatively working towards standardization of laboratory accreditations throughout the industry; ILAC is the international counterpart collaboration of laboratory and inspection accreditation bodies. Established to ensure that laboratories are subject to oversight by an authoritative body, accreditation organizations have been evaluated by peers as competent and have signed arrangements to enhance the acceptance of products and services. Because interested parties can have confidence in the test results and certifications provided by accredited laboratories, the Agency is proposing to amend the laboratory requirement and believes that inexperience with a specific methodology should not disqualify a laboratory from consideration. However, the Agency reserves the right to not accept particular lab data should EPA find cause to doubt the quality and integrity of the work.

(11) *Estimated annual product production volume, average and maximum amounts that*

could be produced per day, and time frame needed to reach maximum production rate (days). While there is currently no requirement for production capability information, the Agency believes it is important for the OSCs and responders to have this information. The availability of a product may impact decisions of authorization of use, depending on inventory or production capabilities. This would prove to be of key importance, for example, in the event of a major environmental disaster (e.g. a SONS event).

(12) *Recognition received from EPA's Design for the Environment (DfE).* EPA's Design for the Environment (DfE)[68] works in partnership with industry, environmental groups, and academia to reduce risk to people and the environment by finding ways to prevent pollution. DfE evaluates human health and environmental concerns associated with traditional and alternative chemicals and processes in a range of industries in order to select safer chemicals and technologies. DfE focuses its review of formulation ingredients on key environmental and human health characteristics of concern within functional classes. This allows formulators to use those ingredients with the lowest hazard in their functional class, while still formulating high-performing products. The DfE label means that EPA scientists have evaluated every ingredient in the product to ensure it meets DfE's stringent criteria. Products that meet Design for the Environment criteria are safer chemical choices. In an effort to encourage the development and use of safer technologies, the Agency is proposing a requirement for the submitter to identify products that have met and can be labeled DfE certified as part of the general information, and would include this information in the NCP Product Schedule Technical Notebook.[69] This Technical Notebook presents manufacturer's summary information on the conditions under which each product is recommended to be used, and is a source of information for the OSC in the event of a response.

[68] http://www.epa.gov/dfe/
[69] http://www.epa.gov/oem/docs/oil/ncp/notebook.pdf

(13) *International product certifications, testing or use data informing the performance capabilities or environmental benefits of the product.* The Agency believes that any additional data available from other countries may help identify the benefits or concerns for the listing and/or the authorization of use of a product. The Agency, however, is not proposing any specific listing criterion or threshold associated with this broad information request, as some products (particularly new formulations) may not have any of this additional data available.

b. Dispersant Testing and Listing Requirements - §300.915(b)

Dispersants are mixtures of solvents, surfactants, and additives that reduce the oil-water interfacial tension and promote the formation of small droplets when mixed with oil. These droplets are driven into the water column by wave action. Emergency response personnel need to know whether a dispersant or any other type of chemical or biological agent on the Schedule could have negative environmental impacts relative to the oil before decisions are made about its use in a particular oil discharge situation. Consequently, it is essential to consider comparative information about the effectiveness and the toxicity of these products. The Agency is proposing revisions to the effectiveness and toxicity testing protocols, as well as to the listing thresholds for dispersants. The proposed revisions are in response to concerns not only for an increase in the frequency of planning for the use of these agents, but also for their potential use in large quantities, such as when responding to oil discharges from oil tanker accidents and offshore well blowouts, as evidenced during the Deepwater Horizon incident in 2010.

(1) Dispersant Efficacy

Current Requirements: The current NCP Subpart J requires dispersants to be tested for saltwater effectiveness in order to determine listing eligibility on the Schedule. Dispersant efficacy is tested following the Swirling Flask Test (SFT) listed in Appendix C to part 300 of the NCP. This

protocol was developed by Environment Canada to provide a relatively rapid and simple testing procedure for evaluating dispersant effectiveness (i.e., the percentage of oil that is dispersed). The procedure places seawater, oil, and a dispersant into an Erlenmeyer flask, which is then placed on a shaker table for a specified period of time. The flask is modified by the addition of a side spout attached to the bottom of the flask, which allows the analyst to pour off dispersed oil from the bottom into a collection vessel for extraction and measurement. After 20 minutes, a sample of water is poured off and chemically analyzed for dispersed oil. For products to be listed they must attain an effectiveness value of 45 percent or greater average dispersion efficacy of two different crude oils (South Louisiana Crude (SLC) and Prudhoe Bay Crude (PBC)) at room temperature (20-23 degrees Celsius (°C)).

Proposed Revisions: The Agency proposes to change the testing protocol for measuring efficacy and adopt the Baffled Flask Test (BFT) as the testing protocol for dispersant efficacy. This protocol has undergone extensive peer review and has been tested in several laboratories, providing reproducible and repeatable results. The Agency also proposes revisions to the efficacy listing criteria for dispersants to be listed. Specifically, the dispersant must demonstrate that the lower 95% confidence level (LCL_{95}) of six replicate flasks meets the new proposed effectiveness listing criteria at two test temperatures. Two new test oils are proposed for this protocol: Alaska North Slope (ANS) and Intermediate Fuel Oil (IFO-120). Both oils are to be tested at two temperatures: 5 °C and 25 °C. Finally, the Agency proposes that dispersants considered for listing must attain the following dispersant effectiveness (DE) values:

- ANS at 5 °C: $DE_{LCL95} \geq 70\%$
- ANS at 25 °C: $DE_{LCL95} \geq 75\%$
- IFO-120 at 5 °C: $DE_{LCL95} \geq 55\%$

- IFO-120 at 25 °C: $DE_{LCL95} \geq 65\%$.

The Agency tested eight of the dispersants currently listed on the Schedule. Three of the eight dispersants clearly differentiated themselves from the other five dispersants for having the best DE_{LCL95} efficacy results; they would meet the thresholds proposed today. For the next best dispersant tested, the results were substantially lower than the third best in the top group. This natural break in the results provided the Agency with the basis for the threshold criteria proposed today. The proposal changes both oils currently used for efficacy testing, replacing SLC and PBC oils with ANS and IFO-120 oils. In proposing two new oils for efficacy testing, the Agency is seeking to ensure that the efficacy testing uses oils that represent a wider range of oil gravities throughout the different geographic locations in the country. The oils selected should exhibit sufficiently different characteristics to produce results that have statistically significant variation between them. The proposal replaces the SLC and PBC oils, both of which were light oils, with the light-to-medium gravity ANS pipeline blend and the heavier refined product, IFO-120. The previous protocol, testing 4 replicates of both the SLC and the PBC oils and combining their results, provided one threshold criterion after determining that each of the oils produced statistically similar results. In contrast, the efficacy test results between the ANS and the IFO-120 showed statistically significant differences at each temperature. The Agency believes that these proposed choices of ANS and IFO-120 represent a wider range of oil gravities and will provide better information on the effectiveness of the products when used on different types of oils.

The Agency is proposing that efficacy testing be conducted at two different temperatures, 5 °C and 25 °C, rather than at an ambient temperature range of 20-23 °C as currently required. The Agency recognizes the current interest in arctic and deepwater drilling, and the continued oil production in the southern, more tropical areas of the country. Given the potential range of

locations where dispersants may be used, the Agency believes it is appropriate to have products tested at temperatures that would reflect that range. These temperatures are intended to capture dispersant use scenarios in a wide range of geographic locations and under different temperatures that may occur in the same geographical location (such as, for example, the deep sea and surface water in the Gulf of Mexico, where the temperatures are typically 5 and 25 °C, respectively).

Further changes to the dispersant testing protocols are designed to improve test repeatability and reproducibility within and between laboratories. The proposed BFT greatly reduces both the inherent error of the method and the human error associated with the current SFT, as discussed below. In addition, reporting the test results in terms of the product's LCL_{95} accounts for between- and within-laboratory error variability and the inherent error of the method. Only one number is reported compared to a mean and standard deviation, as the variation has already been subtracted in the reported number. Specifically:

- The new baffled trypsinizing flask design, fitted with a glass stopcock positioned at the bottom side, promotes less manipulation that could result in erroneous re-suspension of non-dispersed oil, as in the SFT. The re-suspension of oil when using the SFT test protocol was a major source of error. In the SFT, the flask has a side arm spout, requiring the analyst to pick up the flask and pour the contents out manually. Through that action, re-suspension of the dispersed oil can easily take place, leading to potentially erroneous results. In contrast, in the BFT, the flask contains a stopcock at the bottom, so the analyst need only open the stopcock to drain the contents from the flask without the potential of remixing or re-suspending the dispersed oil.
- The BFT provides more turbulent mixing and better enables more reproducible and repeatable dispersant action than the SFT. The mixing energy within the flask is higher

and, as a result of this increased mixing energy, better dispersion is possible. The BFT is also more representative of moderately turbulent sea conditions where dispersants are more likely to be successful when used. Turbulence is needed for the proper mixing of dispersant and oil and for producing the sheer forces needed to create the small droplet sizes with high surface area-to-volume ratio that promote the effective dispersion of the oil into the water column. The BFT provides such mixing and better enables more repeatable and reproducible dispersant effectiveness than the SFT. The BFT was tested extensively in a nine-participant round-robin inter-laboratory calibration test on six commercial dispersant products.[70] In addition, based on the mixing energy that can be achieved using the new baffled flask, the Agency believes a higher effectiveness threshold is warranted. As a result of this increased mixing energy, better dispersion is realized under conditions more realistic of wave action in the sea.

- The variability both between analysts and within analysts is substantially lower for the BFT compared to the SFT, as measured by the coefficient of variation (CV). The CV is defined as the standard deviation divided by the mean and is expressed in terms of percent; the higher the CV, the greater the variability.[71,72]

The Agency considered establishing listing thresholds for efficacy based on the types of oil. The Schedule could list sub-categories of dispersants based on gravities of the oil (i.e., light-medium or heavy), so that a dispersant could only be authorized for use on oils that fall within the range for which it tested effective. Similarly, the Schedule could list dispersants based on

[70] Venosa, A. D., D. W. King, and G. A. Sorial. 2002. The Baffled Flask Test for Dispersant Effectiveness: A Round Robin Evaluation of Reproducibility and Repeatability. Spill Sci. & Technol. Bulletin 7(5-6):299-308.
[71] Sorial, G. A., A. D. Venosa, K. M. Miller, E. Holder, and D. W. King. 2004a. Oil Spill Dispersant Effectiveness Protocol - Part I Impact of Operational Variables. ASCE. J. Env. Eng. Div., 130(10):1073-1084.
[72] Sorial, G. A., A. D. Venosa, K. M. Miller, E. Holder, and D. W. King. 2004b. Oil Spill Dispersant Effectiveness Protocol - Part II Performance of the Revised Protocol. ASCE ASCE. J. Env. Eng. Div., 130(10):1085-1093.

temperature, allowing a dispersant to be authorized for use only at the temperature for which it tested effective. However, determining ranges or definitions for when oil changes from light to medium or heavy, is challenging because they may be affected by a number of external factors, including temperature. Likewise, listing dispersants based on temperature would necessitate an additional degree of complexity to the authorization for use, as the temperatures may change drastically in some geographical regions affected by the same oil discharge. Even within more stable geographical regions, there may be variations at the time of an oil discharge that could affect the decision to use a dispersant stockpiled for a different temperature range use. While these more specific listing approaches would provide more information on effective product choices, the Agency is concerned about how such specific listing approaches may contribute to OSC confusion in an emergency, the cost of stockpiling multiple types of dispersants, or the immediate availability of the appropriate dispersant in an emergency. Furthermore, the Agency is proposing revisions to Subpart J to authorize RRTs to request supplementary testing and information to provide greater flexibility in tailoring testing conditions to address area- and site specific concerns. Given these factors, the Agency believes that it is appropriate that only those dispersants that achieve effectiveness for both types of oils at both temperatures be listed. Listing only the products that achieve effectiveness for both types of oils at both temperatures would demonstrate a wide range of effectiveness for the product, and a single listing would facilitate authorizing their use during a response action.

(2) Dispersant Toxicity

Current Requirements: The current NCP Subpart J requires dispersants to be tested for toxicity to saltwater species. This information is available to the OSC for their consideration in authorizing dispersant use. Dispersant toxicity is tested following the protocols in Appendix C to

part 300 of the NCP. The standard toxicity test for dispersants involves exposing two saltwater species, the inland silverside fish (*Menidia beryllina or M. beryllina*) and the mysid shrimp (*Americamysis bahia or A. bahia*), to five concentrations of the test product and No. 2 fuel oil, alone and in a 1:10 mixture of product to oil. Reference toxicity tests are conducted using dodecyl sodium sulfate (DSS) as a reference toxicant. The test length is 96 hours for *M. beryllina* and 48 hours for *A. bahia*. The concentration of test product causing 50% lethality to the test organisms (LC_{50}) is calculated at the end of the exposure period.

Proposed Revisions: The Agency proposes to revise the toxicity testing requirements for dispersants, including the testing protocol and the use of the test results. The proposal would require acute toxicity testing for the dispersant alone, the dispersant mixed with ANS, and the dispersant mixed with IFO-120. While the Agency currently provides the toxicity testing results to the OSC to assist in authorization of use determinations, it is proposing to use these testing results to determine eligibility for listing on the Schedule.

Specifically, the Agency is proposing to require evaluations of dispersant toxicity using acute toxicity tests following existing USEPA effluent testing guidelines,[73,74] modified for use with dispersant product or dispersant product/oil mixtures, as detailed in Appendix C to part 300. Measured toxicity values are proposed to include static acute toxicity tests with dispersant product and dispersant product/oil mixtures using *A. bahia* and *M. beryllina*. Developmental toxicity using a sea urchin assay and sub-chronic effects using *A. bahia* and *M. beryllina* are proposed to be conducted with dispersant product only. The Agency is proposing the following toxicity thresholds

[73] USEPA 2002a Methods for Measuring the Acute Toxicity of Effluents and Receiving Waters to Freshwater and Marine Organisms, Fifth edition. (Acute Methods Manual), Office of Water, Cincinnati, OH. EPA-821-R-02-012.
[74] USEPA 2002b. Short-term Methods for Estimating the Chronic toxicity of Effluents and Receiving Waters to Marine and Estuarine Organisms, Third Edition. (Saltwater Chronic Methods manual). Office of Water, Cincinnati, OH. EPA-821-R-02-014.

for listing dispersants on the Schedule:

- The lower bound of the LC_{50} 95% confidence interval (CI) greater than 10 ppm for acute toxicity;
- The lower bound of the IC_{50} 95% confidence interval greater than 10 ppm for developmental toxicity;
- No Observed Effect Concentration (NOEC) equal to or greater than 1 ppm for sub-chronic toxicity.

Test Oils and Oil Acute Toxicity Values. The Agency proposes to replace the No. 2 fuel oil currently used for toxicity testing with two test oils, ANS and IFO-120. In addition to making the test oils consistent with the dispersant efficacy tests, testing oils of different gravities also provides a better estimation of dispersant /oil toxicity associated with differing oil constituencies, and ultimately a better representation of the potential overall toxicity of a product.

The Agency will conduct acute toxicity tests using the two reference oils with both *A. bahia* and *M. beryllina* and will make all test results available to the public, including raw data and a calculated median LC_{50} values, on the EPA website. Acute toxicity values for the test oils can be used for comparisons with dispersant-oil mixture acute tests. By providing this information, the Agency is reducing the number of required toxicity tests the submitter would need to conduct. While requiring submitters to conduct oil-only acute toxicity testing could provide an opportunity to detect anomalies in the submitted data, the Agency is not proposing this as a requirement. The Agency is requesting comments on whether the submitter should be required to conduct the acute toxicity testing for the two test oils.

Dispersant-Only Acute Toxicity. Using the EPA toxicity classification scheme[75], LC_{50} values

[75] USEPA 2010. http/www.epa.gov/oppefed1/ecorisk_ders/toera_analysis eco.htm#Ecotox.

ranging from 10 ppm to 100 ppm are classified as slightly toxic and above 100 ppm substances are considered acutely nontoxic to aquatic organisms. For both *M. beryllina* and *A. bahia*, the Agency is proposing to use the lower bound of the LC_{50} 95% confidence interval (CI) as the decision point and is proposing to set this threshold value to be greater than 10 ppm for a dispersant to be listed on the Schedule. The rationale for using the lower bound of the CI is that the confidence interval should not contain any values less or equal to 10 ppm since theoretically, the LC_{50} can fall anywhere within the CI. By using the lower CI the Agency is providing a conservative decision criterion for acute toxicity, and by proposing a greater than 10 ppm threshold level it is establishing an adequate safety margin without being overly restrictive.

Dispersant-Oil Mixture Acute Toxicity. The Agency is proposing to substitute the existing preparation procedure for the dispersant-oil mixture (DOM) in Appendix C to Part 300, Section 3 method for the preparation procedure for dispersant-oil mixture using a modified protocol[76] first described by Baron and Ka'aihue[77] and fully described by Hemmer et al.[78] These new modified procedures for preparation of the chemically enhanced water accommodated fractions (CEWAFs) of dispersant-oil mixtures are used to make exposure solutions. Evaluations of the dispersant-oil mixture toxicity using acute toxicity tests follow existing EPA effluent testing guidelines,[79,80] modified for use with these exposure solutions and are detailed in Appendix C to part 300. EPA

[76] The modified protocol is based on a variable dilution modification of the Chemical Response to Oil Spills: Ecological Research Forum (CROSERF) testing protocol; Aurand, D. and G. Coelho (Editors). 2005. Cooperative Aquatic Toxicity Testing of Dispersed Oil and the "Chemical Response to Oil Spills: Ecological Effects Research Forum (CROSERF)." Ecosystem Management & Associates, Inc. Lusby, MD. Technical Report 07-03; http://www.ecosystem-management.net/assets/documents//FINAL%20CROSERF%20REPORT.pdf

[77] Barron, MG, Ka'aihue L. 2003. Critical evaluation of CROSERF test methods for oil dispersant toxicity testing under subarctic conditions. Mar Pollut Bull 46:1191-1199.

[78] Hemmer, MJ, Barron, MG, Greene, R. 2011. Comparative toxicity of eight oil dispersants, Louisiana sweet crude oil (LSC) and chemically dispersed LSC to two aquatic species. Environ Toxicol Chem, 30: 2244–2252

[79] USEPA 2002a Methods for Measuring the Acute Toxicity of Effluents and Receiving Waters to Freshwater and Marine Organisms, Fifth edition. (Acute Methods Manual), Office of Water, Cincinnati, OH. EPA-821-R-02-012.

[80] USEPA 2002b. Short-term Methods for Estimating the Chronic toxicity of Effluents and Receiving Waters to Marine and Estuarine Organisms, Third Edition. (Saltwater Chronic Methods manual). Office of Water, Cincinnati, OH. EPA-821-R-02-014.

proposes to modify the variable loading preparation procedure described in CROSERF to a variable dilution procedure to standardize the oil-to-water ratio and provide a consistent chemical concentration in a test series. Additionally, the modified preparation procedure provides economies in analytical costs by reducing the need to analyze the composition of every tested concentration. Chemical analysis for the total petroleum hydrocarbon (TPH) concentration of the CEWAF stock solution is required.

Sea Urchin Developmental Assay. The Agency is proposing to require a sea urchin developmental assay (also referred to as an embryogenesis assay) to assess the potential for a dispersant product to cause adverse effects on the developmental process. During the Deep Water Horizon spill event, the possible adverse effects of oil dispersants on the developmental processes of fish and invertebrate species were identified as critical issues of concern. By incorporating the sea urchin developmental assay, the Agency is using it as a sensitive surrogate test for pelagic early life stages. This assay would employ an existing EPA test protocol[81] routinely used in effluent testing. As suggested for the dispersant acute toxicity test criteria with *A. bahia* and *M. beryllina*, the Agency is proposing to use the lower bound of the IC_{50} 95% confidence interval as the decision point and the threshold value to be greater than 10 ppm. For this test, the IC_{50} is defined as the concentration of dispersant that inhibits the development of 50% of exposed embryos. The Agency believes that a greater than 10 ppm threshold level provides an adequate safety margin without being overly restrictive.

Sublethal/Subchronic Studies. The Agency has limited information concerning the possible sublethal effects of dispersants currently listed on the Product Schedule. The Agency is proposing requirements for subchronic assays (duration approximately 7 days) to be performed with *A. bahia*

[81] USEPA 1996. Short-term Methods for Estimating the Chronic Toxicity of Effluents and Receiving Waters to West Coast Marine and Estuarine Organisms. EPA/600/R-95/136

and *M. beryllina* following established EPA short-term methods for estimating chronic toxicity of effluents.[82] While the Agency considered requiring longer-term tests, it believes the 7-day subchronic tests are reasonable alternatives for estimating chronic toxicity. This information would also be of value as guidance to regional responders and OSC's on possible adverse effects on survival and growth of larval fish and invertebrates caused by longer-term exposure to dispersants. The Agency is proposing a NOEC equal to or greater than 1 ppm listing threshold level for sub-chronic effects, and believes it provides an adequate safety margin without being overly restrictive. The Agency is requesting comment on whether it should consider longer-term tests (e.g., early life stage tests), which may be more relevant to sublethal effects caused by longer duration exposure to low concentrations of a dispersant product.

Agency Review of Test Results. While all proposed toxicity tests would be required for Schedule listing determinations, a systematic approach to testing products may prove useful and cost effective for submitters. The Agency recommends looking first for acceptable efficacy testing values and then measuring those against the listing threshold criteria for: dispersant acute toxicity, sea urchin developmental assay, acute toxicity testing of dispersant-oil mixtures, and finally sublethal/chronic studies. This product testing approach would provide an efficient pathway for evaluating the efficacy and toxicity testing of dispersant products. Following a similar approach in product development and testing may help identify unsuitable products early in the evaluation process.

Supplemental Options:

In addition to the criteria discussed above, the Agency considered calculating a value based

[82] USEPA 2002b. Short-term Methods for Estimating the Chronic toxicity of Effluents and Receiving Waters to Marine and Estuarine Organisms, Third Edition. (Saltwater Chronic Methods manual). Office of Water, Cincinnati, OH. EPA-821-R-02-014.

on the ratio of the LC_{50} of oil-alone to the LC_{50} of the dispersant-oil mixture for *M. beryllina* and *A. bahia* to assist in comparing the relative toxicity of crude oil to chemically dispersed oil. A ratio of less than or equal to 1 would indicate the toxicity of the dispersant-oil mixture is less than or equal to the toxicity of the oil-alone. The Agency considered whether this ratio should be used as a criterion for inclusion of dispersants on the Schedule or as guidance to emergency responders. Using a less than or equal to 1 ratio as a guideline would afford an adequate safety margin without being overly restrictive. While the ratio would provide a simple, interpretable, and easily justifiable value for determining acceptable toxicity levels of dispersant-oil mixtures, it would not be specific to oils in particular discharge situations. The Agency is requesting comment as to whether such a value should be calculated, and if so, whether it should be used as an additional listing criterion or provided as information for the responders to consider in authorizing dispersant use.

The Agency also considered including geographically or ecologically representative species in the testing protocol. The Agency believes, however, that this issue is addressed by the proposed revisions to the supplemental testing and information requirements in section 300.910 (g), with the decisions of testing geographically and ecologically representative species left to the discretion of the RRTs. While inclusion of species from different phyla and habitats would provide useful and important information on possible adverse effects of dispersant products and dispersed oil, the proposed testing protocols would need to be modified. Further, the Agency is concerned about balancing the time and cost associated with the development of these tests on the part of the submitter rather than on the end users.

The Schedule currently requires no specific toxicity or efficacy tests for subsea dispersant listing or authorization of use. While the differences in toxicity values and efficacy may be affected by application in subsea environments, the Agency believes that the requirements proposed today

establish an adequate baseline for listing dispersants on the Schedule and for authorizing their use by responders in case of an incident. The Agency is addressing subsea concerns at this time by revising the authorization of use requirements for agents to include new provisions for subsea monitoring, and requests comments on alternative testing and listing approaches to specifically address subsea concerns.

(3) Limitations of Use

The Agency is proposing a conditional use listing for dispersants. The proposal specifies that dispersants may only be used in saltwater. Dispersants are typically designed and traditionally used for responding to oil discharges in saltwater. In general, the effectiveness of dispersants decreases as the salinity of the water decreases, given the same hydrophilic-lipophilic balance, and the effectiveness is minimal in freshwater environments. Surfactants, the main active ingredient in dispersants, sustain oil droplets in water by orienting the lipophilic side of its molecule in the oil and the hydrophilic side in the water. The hydrophilic side of the surfactant is generally more soluble in waters with higher salinity values, causing the oil droplets to be more stable in saline water environments. In waters with no salinity, many dispersants have shown a very low effectiveness or are sometimes completely ineffective.[83]

The Agency is also concerned with using dispersants in freshwater environments because of the limited dilution typically available as compared with the open sea and because of the existence of water intakes in rivers, streams, and lakes for use in drinking water supplies. Using these agents in freshwater has the potential for compounding the impacts caused by already discharged petroleum products, particularly near potable and non-potable subsurface water intakes. The Agency does recognize, however, that as a last resort, such agents may have some limited

[83] Fingas, M., (Ed.), 2011, *Oil Spill Science and Technology*, Gulf Professional Publishing, pp 513-518

applicability in freshwater. The provision for authorization of use under §300.910(d) allows for exceptions to prohibition on the use of dispersants in freshwater in cases where there is an immediate threat to human life. The Agency is requesting comment on this proposed listing limitation.

c. Surface Washing Agent Testing and Listing Requirements - §300.915(c)

Surface washing agents are substances composed primarily of surfactants and/or solvents that remove oil from solid surfaces, such as beaches, rocks, or concrete, through a detergency mechanism that lifts and floats the oil for collection and removal from the environment with minimal dissolution, dispersion, or transfer of oil into the water column. For this agent category, the Agency is revising the toxicity testing protocols, adding effectiveness testing protocols, and establishing both toxicity and efficacy listing thresholds. The proposed revisions respond to concerns regarding their frequent use and the potential for residual impacts after their use.

(1) Surface Washing Agent Efficacy

Current Requirements: There are currently no efficacy testing requirements in the NCP Subpart J for surface washing agents to determine listing eligibility on the Schedule.

Proposed Revisions:

The Agency is developing a laboratory testing protocol to evaluate the effectiveness of SWAs in removing crude oil from a solid substrate.[84] Currently available effectiveness testing protocols for these agents are concerned with assessing the detergency, or cleaning power, of the agents by quantifying the physical removal of oil from non-vegetative substrates as a result of their

[84] Development and testing of a new protocol for evaluating the effectiveness of oil spill surface washing agents. Karen Miller Koran, Pegasus Technical Services, Inc., 46 E. Hollister Street, Cincinnati, OH 45219; Albert D. Venosa, US Environmental Protection Agency, 26 W. Martin Luther King Drive, Cincinnati, OH 45268,*, Christopher C. Luedeker, Keith Dunnigan StatKing Consulting, Inc., 759 Wessel Drive, Suite 6, Fairfield, Ohio 45014; George A. Sorial, University of Cincinnati, P.O. Box 210071, Cincinnati, OH 45221-0071. Marine Pollution Bulletin. journal homepage: www.elsevier.com/locate/marpolbul

use. The protocols typically involve the application of oil to a solid substrate (or the use of pre-oiled substrate), weathering of the oil on the substrate, application of the agent, a contact period for SWA penetration, and washing of the substrate with water. The fractions of oil removed in the wash water and/or remaining on the substrate are quantified. Oil removal efficiencies for surface washing treatments are typically compared to the washing efficiency of water without surface washing agents (untreated controls). Disadvantages or potential sources of error have been identified in existing protocols that may not reflect real world performance. Of concern, for example, the use of substrates (e.g., stainless steel, porcelain) with different wetting and adhesion properties than natural substrates such as sand or gravel; short weathering times insufficient for oil attachment; and the absence of mixing energy during the washing step may lead to incomplete detachment of partially released oil. The Agency's goal is to develop and evaluate a new bench-scale testing protocol that would provide a standard for EPA to use in surface washing agent product evaluation prior to listing on the Schedule. Changes over established protocols the Agency considered include: using natural substrates to reflect the most commonly impacted shoreline material; establishing sufficient weathering time to permit strong attachment of the oil to the substrate surface; fully quantifiable approaches with known volumes and precise measurements to allow mass balance; and a standardized and reproducible methodology to minimize operator error. PBC oil was used as the representative standard reference oil in developing the protocol since it had previously been selected as one of two standard oils for the Agency's published protocols for dispersant and bioremediation agent testing. However, the Agency is proposing revisions to the dispersant and bioremediation testing protocols, including replacing the PBC reference oil. The protocol being developed has not been tested: with the IFO-120 or the ANS oils that are proposed to be used in the revised dispersant effectiveness testing protocol; at multiple temperatures; or to include freshwater

systems. The Agency expects to develop final recommendations for the testing protocol following round robin testing after adjusting for the new variables.

Because of the additional work needed to finalize its protocol, the Agency at this time is proposing to allow the use of standard recognized efficacy testing methodologies for surface washing agents. An example of such a standard recognized methodology is the American Society for Testing and Materials (ASTM) Standard Test Method for Evaluating the Effectiveness of Cleaning Agents.[85] Another methodology is Environment Canada's Test Method.[86] The capability of a particular surface washing agent depends upon the application procedures and the characteristics of the surface being cleaned, such as size, shape, and material. The ASTM test method in particular covers a procedure for evaluating the capability of the agents, providing a relatively rough surface to which the oil can adhere. The Environment Canada method uses a stainless-steel 'trough' which is placed at a specified angle. The target oil, is placed on an area on the trough. The treating agent is applied in droplets to the surface of the oil and after 10 minutes at 5 minute intervals, rinses of water are applied to the trough. After drying the trough is weighed and the removal calculated from the weight loss. Repeatability is within 5%. The Agency is requesting comment on available methodologies and its published protocol to incorporate as the testing protocol for these agents.

The Agency is also proposing an efficacy threshold of 30% in either fresh or salt water or both depending on the intended product use. This is based on the efficacy data published in "Surface-washing Agents or Beach Cleaners" (see footnote 86). Numerous surface washing agent

[85] ASTM Standard Test Method for Evaluating the Effectiveness of Cleaning Agents. Designation: G122 – 96 (Reapproved 2008). ASTM International, 100 Barr Harbour Dr., PO box C-700 West Conshohocken, Pennsylvania 19428-2959, United States

[86] Fingas, Merv and Fieldhouse, Ben; "Surface-washing Agents or Beach Cleaners" (2010). Chapter 21 Surface-Washing Agents or Beach Cleaners. In Oil Spill Science and Technology (p716). London: Gulf Professional Publishing.

products including several from the Schedule were tested using the Environment Canada methodology. The average efficacy of the surface washing agents tested is approximately 30%. The Agency requests comments on this approach and other efficacy test data available.

(2) Surface Washing Agent Toxicity

Current Requirements: The current NCP Subpart J requires surface washing agents to be tested for saltwater species toxicity. Surface washing agent toxicity is tested following the dispersant toxicity testing protocols in Appendix C to part 300 of the NCP.

Proposed Revisions: The Agency is proposing revisions to the toxicity testing requirements for surface washing agents, including the testing protocol. While the Agency currently provides the toxicity testing results to the OSC to assist in authorization of use determinations, it is proposing to use these toxicity testing results to determine listing eligibility on the Schedule.

The proposed revisions to the testing protocols for surface washing agents are detailed in Appendix C to part 300. The standard acute toxicity test protocol for surface washing agents is based on EPA's protocol, *Methods for Measuring the Acute Toxicity of Effluents and Receiving Waters for Freshwater and Marine Organisms.* The Agency proposes to require surface washing agents be tested for acute toxicity using fresh water species *Ceriodaphnia dubia* and *Pimephales promelas*, or saltwater species *Americamysis bahia* and *Menidia beryllina*, or both, depending on the intended product use. The concentration of test product causing 50% lethality to the test organisms (LC_{50}) and lower and upper 95% confidence intervals (LCI_{95} and $ULCI_{95}$) are calculated at the end of the exposure period. For the proposed listing threshold, a test product must demonstrate an acute LCI_{95} greater than 10 ppm in either fresh or salt water, consistent with the acute toxicity thresholds proposed for dispersants. Using the EPA toxicity classification scheme,[87]

[87] USEPA 2010. http/www.epa.gov/oppefed1/ecorisk_ders/toera_analysis eco.htm#Ecotox.

LC_{50} values ranging from 10 ppm to 100 ppm are classified as slightly toxic and above 100 ppm substances are considered practically nontoxic to aquatic organisms. By proposing an LCI_{95} greater than 10 ppm threshold level, the Agency is establishing an adequate safety margin without being overly restrictive.

(3) Limitations

The Agency is proposing a conditional use listing for surface washing agents. The proposal specifies that these products may only be used in those waters for which the product was tested and for which it met the listing threshold criteria. The Agency recognizes that products may yield effective results in certain environments and not in others. Products that may be effective in freshwater may not necessarily be so in saltwater, and vice versa. The Agency is proposing this limitation as a flexible alternative to product manufacturers, who may choose to test a product for a selected environment. The product will be listed and may only be authorized for use in those water environments for which it was tested and for which it met both the efficacy and toxicity listing criteria.

d. **Bioremediation Testing and Listing Requirements** §300.915(d)

The Agency is proposing to define bioremediation agents as microorganisms, biologically derived compounds such as enzymes, or nutrients deliberately introduced into a contaminated environment to increase the rate of biodegradation and mitigate any deleterious effects caused by the contaminant constituents. Bioremediation agent categories include, but are not limited to, biological agents, enzymes, and nutrients (i.e., fertilizers) containing bioavailable forms of nitrogen and phosphorus. For this agent category, the Agency is proposing revisions to the efficacy testing protocols, adding toxicity testing protocols, and establishing both efficacy and toxicity listing thresholds.

(1) Bioremediation Agent Efficacy

Current Requirements: Bioremediation agent efficacy is currently tested following the protocols in Appendix C to part 300 of the NCP. The current bioremediation agent effectiveness testing protocol is designed to determine a product's ability to biodegrade oil by quantifying changes in the oil composition resulting from biodegradation using natural seawater as the test water. The protocol tests the bioremediation agent for microbial activity and quantifies the disappearance of saturated hydrocarbons and polynuclear aromatic hydrocarbons (PAHs) in weathered ANS 521.[88] The sample preparation procedure extracts the oil phase into dichloromethane (DCM), with a subsequent solvent exchange into hexane. The analytical technique uses a high-resolution gas chromatograph/mass spectrometer (GC/MS); GC/MS provides for a high degree of chemical separation and spectral resolution. Concurrently with the chemical analysis, a microbiological analysis is also performed to determine and monitor the viability and growth of the microbial cultures being studied. For commercial products that are strictly microbial agents not containing their own nutrients, a mineral nutrient solution is provided if requested by the submitter; no further nutrients are added to commercial products containing their own nutrients. To be listed on the Schedule, bioremediation agents must demonstrate a statistically significant difference between the average degradation of both the total alkanes and the total aromatics in the oil achieved by the product after 28 days and the average degradation for the same compounds achieved by the non-nutrient control in the same time period.

Proposed Revisions: The proposal reflects a series of changes to the current efficacy testing protocol for bioremediation agents. The protocol includes freshwater testing in addition to the

[88] Artificially distilled Alaska North Slope crude oil at 521°F (272°C) to remove the low molecular weight hydrocarbons to approximate natural weathering processes that occur after a spill.

current saltwater-based test, and uses artificial water for both fresh and saltwater testing, replacing the natural seawater currently used. The proposed protocol also eliminates several gravimetric and microbiological analyses and testing endpoints not used in listing determinations. Additionally the proposed protocol limits the levels at which external nutrients may be added, which allows the addition for product formulations without nutrients, or for product formulations that have nutrient concentrations at insufficient levels for the experimental setup. Finally, the methodology streamlines the statistical analysis. The proposed revisions address concerns with the existing methodology, expanding its application to include freshwater environments, improving the consistency and comparability of the test results, and generally streamlining the protocol.

Protocol. The proposed efficacy testing protocol is designed to determine a bioremediation agent's ability to biodegrade weathered crude oil in the exposure water (saltwater or freshwater) by quantifying changes in the oil composition resulting from biodegradation. It consists of a series of six product and control flasks containing artificial seawater or artificial freshwater and weathered crude oil in which biodegradation of the crude oil hydrocarbons is monitored for 28 days. Product flasks at day 28 are compared to product flasks at day 0 to determine if a specified reduction both in total resolved alkanes and total resolved aromatics was achieved in that time period. Product flasks contain exposure water, weathered oil, and product in concentrations specified by the manufacturer. Positive control flasks must contain exposure water, oil, nutrients, and the standard culture supplied by the Agency. A negative control, consisting of artificial exposure water, product, weathered crude oil, nutrients, and a sterilant is also used to ensure that the observed degradation of hydrocarbons was not caused by abiotic losses or interaction with the product. The proposed protocol quantifies the degradation (or disappearance) of alkane hydrocarbons and aromatic hydrocarbons. The sample preparation procedure extracts the oil phase into the solvent

dichloromethane (DCM; also known as methylene chloride) with a subsequent solvent exchange into hexane. The test method targets the relatively easy to degrade normal alkanes and the more resistant and toxic aromatics. The analytical technique uses a high resolution GC/MS because of its high degree of chemical separation and spectral resolution. GC/MS has long been used to study the weathering and fate of oil spilled into the environment. It gives unambiguous results for determining biodegradation effectiveness. For quantitative analyses, the instrument is operated in the selected ion monitoring mode (SIM) of detection at a scan rate of > 5 scans per second to maximize the linear quantitative range and precision of the instrument.

Fresh and Saltwater Environments. The existing bioremediation test is exclusively for saline environments with no equivalent test for freshwater environments. Further, the existing test uses natural seawater and has been found to give variable results due to the compositional variability of seawater both chemically and microbiologically. The Agency proposes to replace the natural seawater with a standardized artificial saltwater formula called GP2, whose components and concentrations are generally recognized, and which is easily made. The use of artificial saltwater allows for better test reproducibility. Additionally, the Agency proposes to expand the requirement for efficacy testing to include freshwater, which allows for a better evaluation of the use of these agents in this environment. Similarly to the artificial saltwater, artificial minimal salts freshwater would be used with known ingredients (Bushnell and Haas; Journal of Bacteriology, Vol. 41: 653, 1941). Because the Agency is adding a freshwater test, it is also allowing the submitter to test a product for freshwater only, saltwater only, or both. The Schedule listing would specify in which environment the product was tested and authorized for use. Products could be listed for use on saltwater only, on freshwater only, or for use on both environments, giving the option to the submitter to market its products accordingly.

GC/MS Testing Endpoints. GC/MS has long been used to study the weathering and fate of oil spilled into the environment. However, GC/MS being the primary tool used in the analysis makes this test expensive. The current protocol includes several measured variables in both the product flasks and the non-nutrient control flasks, including the gravimetric residual oil weight analysis, a Most Probable Number microbiological analysis, and the GC/MS analysis at 3 different time points, days 0, 7, and 28. The Agency is proposing to eliminate all of these analyses except the day 0 and 28 GC/MS of both the non-nutrient control and the product flasks. The Agency believes this is appropriate because the other measurements are not used in determining whether a product meets the listing criterion. Additionally, the statistical analysis has been greatly simplified, and a new decision rule is proposed for listing determinations rather than relying on a statistical significance test.

- *Gravimetric Weight Analysis:* Originally, the gravimetric weight analysis was used as a preliminary test to avoid having to perform a GC/MS analysis. Products that failed to significantly reduce the gravimetric weight of the oil within 28 days were considered not effective, thereby eliminating the need to conduct the more expensive GC/MS analysis. However, numerous compounds can give rise to positive interferences with the gravimetric weight analysis, such as biomass lipids and other biogenic materials, thereby rendering the gravimetric analysis suspect. Furthermore, numerous factors can confound the interpretation of gravimetric weight changes in oil over time. This has led to confusion as some manufacturers have mistakenly concluded their product passed the testing protocol simply by complying with the gravimetric weight loss measurement. By eliminating the gravimetric analysis altogether, EPA is streamlining the testing process, resulting in less confusion and a modest analytical cost savings for the manufacturer.

- *Most Probable Number Enumeration (Microbiological Analysis), Nutrient Control, And Day-Seven Sampling Event:* Results of these test endpoints, while currently required, are not used in determining whether a product meets the current listing requirements. Therefore, the Agency is proposing to eliminate these test requirements from the proposed protocol.
- *28 day Sampling Event:* Generally, results of biodegradation may occur in less than 28 days, particularly for alkanes for which biodegradation often takes place in a much shorter time period. However, 28 days are needed to make sure that the aromatic components, which biodegrade more slowly, have been reduced significantly; testing protocols that accommodate aromatics are of particular importance because of their potential ecological toxicity. While the Agency could propose different time frames in determining whether the product met the reduction criterion for alkanes and aromatics, it believes that such an approach would be confusing. Therefore, the 28 day time period is being proposed for both fractions for simplicity and to provide sufficient time for degradation of the aromatics.

Protocol Verification. The Agency proposes to provide a standard culture for performing laboratories to use as a positive control benchmark. A positive control is needed as an indication that the test was performed properly. The Agency is proposing to use as the standard culture an oil-degrading bacteria isolated from Disk Island in Prince William Sound in 1990. This culture is an excellent degrader of alkanes and aromatics in saltwater and freshwater, although it performs better in saltwater, especially in degrading the aromatics. This culture has the ability to degrade ANS 521 oil, with known effectiveness values for both fresh and saltwater. The proposed positive control consists of triplicate flasks containing sterile artificial seawater or freshwater, nutrients (salts of nitrogen and phosphorus), the weathered ANS 521 oil, and the standard culture. The performing laboratory must achieve the known reduction in the alkane and aromatic fractions in these positive

control flasks in order to certify they successfully conducted the test procedure. Data from the positive control are also required to be submitted with the product test results. The Agency is not proposing, however, that the positive control be repeated every time a product test is performed. Rather, the certifying laboratory would be required to report the results of the positive control tests conducted within a year prior to the product testing, as part of the product submission package. Additionally, this positive control culture would be used as a standard microbial inoculum to test abiotic products, such as nutrients and enzymes.

Added Nutrients. The current protocol allows the manufacturer to request the addition of nutrients to support a culture's ability to degrade the crude oil hydrocarbons. This addition is currently allowed for commercial products that are strictly microbial agents and do not contain any nutrients; the addition of nutrients is not allowed for commercial products containing its own nutrients. A modified salt solution of nitrogen and phosphorus is used as the mineral nutrient. All microbial cultures need nutrients to survive and grow. In the natural environment, biodegradation is not completely limited by nutrient availability given that a low level flow of nutrients is continuous. To maximize biodegradation rates, the addition of nutrients is typically recommended in the field with a bioremediation product unless nutrients are already high (i.e., > 5 mg/L). However, in a closed laboratory flask, the amount of nitrogen and phosphorus supplied by a typical product might easily become limiting quickly as biodegradation takes place because they are supplied only once at the beginning of the test. If nutrient limitation occurs, further reduction of hydrocarbons would likely be inhibited, and this may cause the product to fail the listing criteria. To address this problem, the Agency proposes modifying the protocol by allowing the addition of nutrients to any product containing living microbes that does not contain said nutrients in sufficient quantities to allow a fair evaluation. However, the additional quantities allowed are limited to total

concentrations of nitrogen and phosphorus that do not to exceed the equivalent amounts used in the positive control. Capping the amount of nutrients that can be added to all products will yield more consistent results and more effective comparisons among products.

Efficacy Thresholds. The Agency is proposing to revise the listing criteria by replacing the current "statistically significant difference" test with defined values for the percent reduction of total alkanes and aromatics concentrations. For a bioremediation agent to be listed on the Schedule, the Agency proposes the percentage reduction of the total alkane fraction (sum of all alkane concentrations determined by GC/MS) to be greater than 95% at day 28, and that the percentage reduction of the total aromatic fraction (sum of all aromatic concentrations determined by GC/MS) must be greater than 70% at day 28, both based on the one-tailed Upper 95% Confidence Limit (UCL_{95}) for salt water. For freshwater, the Agency proposes the percentage reduction of the total alkane fraction (sum of all alkane concentrations determined by GC/MS) to be greater than 95% at day 28, and that the percentage reduction of the total aromatic fraction (sum of all aromatic concentrations determined by GC/MS) must be greater than 40% at day 28, both based on the one-tailed Upper 95% Confidence Limit (UCL_{95}).

Basis of Efficacy Thresholds. According to Haines *et al.* (2005),[89] the positive control culture was able to biodegrade total alkanes in saltwater by 98.9% and total aromatics by 79.8% in 28 days, based on the UCL_{90}. In freshwater, the percent reduction values were 97.9% and 37.8%, respectively, again based on the UCL_{90}. Thus, the proposed listing thresholds for the percentage reduction of total alkanes and total aromatics from the GC/MS analysis are based on the findings of Haines *et al.* (95% for alkanes vs. 98.9% in saltwater and 97.9% in freshwater, 70% for aromatics

[89] Haines, J.R., E.J. Kleiner, K.A. McClellan, K.M. Koran, E.L. Holder, D.W. King, and A.D. Venosa. 2005. "Laboratory evaluation of oil spill bioremediation products in salt and freshwater systems." J. Ind. Micorbiol. Biotechnol. 32:171-185.

in saltwater and 40% in freshwater vs. 79.8% in saltwater and 37.8% in freshwater). As for the positive control, the Agency believes a reasonable target in saltwater would be 95% for total alkanes in both types of exposure waters. For aromatics reduction, 70% is considered reasonable in saltwater and 40% in freshwater, based on Haines *et al.* EPA is using UCL_{95} values rather than the UCL_{90} values from Haines because EPA's latest research uses 6 replicates for the test protocol which increases precision. The proposed criteria demonstrate that the product can cause a substantial degradation of the alkane and aromatic fractions of weathered crude oil compared to a control, as determined solely by GC/MS analysis.

Of note, these proposed thresholds for listing bioremediation products on the Schedule are similar to those established for Canadian efficacy testing, which are reductions of 30% for total aliphatics and 10% for total aromatics. The 30% reduction in total aliphatics (including all resolvable and non-resolvable but GC-detectable aliphatics) is approximately equivalent to an 80% reduction in total GC/MS-resolvable target n-alkanes, based on Canadian data. The 10% reduction in total aromatics is approximately equivalent to a 50% reduction of the 5-aromatic homologue group consisting of naphthalene, fluorene, dibenzothiophene, phenanthrene, and chrysene and their alkylated homologs. The U.S. aromatic series includes 2 other 4-ring aromatics in addition to those 5 aromatic series considered by Blenkinsopp et al.[90]Thus, the U.S. and Canadian protocols are similar in terms of decision criteria. The Agency requests comments on all the proposed changes and listing thresholds.

(2) Bioremediation Agent Toxicity

Current Requirements: The Agency currently has no bioremediation agent toxicity testing

[90] Blenkinsopp, S, G Sergy, Z Wang, MF Fingas, J Foght and DWS Westlake. 1995. Oil spill bioremediation agents-Canadian efficacy test protocols. Proceedings of the 1995 International Oil Spill Conference, Feb 27-March 2, Long Beach, CA. American Petroleum Institute, Washington, D.C. pp. 91-96.

requirements for purposes of listing these agents on the Schedule. Section 5 of Appendix C is reserved for this purpose. The Agency has, however, on a case-by-case basis, requested manufacturers to test bioremediation agents for toxicity if the product contains surfactants or other ingredients that may be harmful to the environment.

Proposed Revisions: The Agency is proposing an acute toxicity testing protocol for bioremediation agents to include both fresh and saltwater. The Agency will use these testing results to determine listing eligibility on the Schedule. The proposed testing protocols for bioremediation agents are detailed in the proposed Appendix C to part 300. The standard acute toxicity test protocol for bioremediation agents is based on EPA's protocol, *Methods for Measuring the Acute Toxicity of Effluents and Receiving Waters for Freshwater and Marine Organisms.* The Agency proposes to require bioremediation agents be tested for acute toxicity for the product alone using fresh water species *Ceriodaphnia dubia* and *Pimephales promelas*, and saltwater species *Americamysis bahia* and *Menidia beryllina*. The concentration of test product causing 50% lethality to the test organisms (LC_{50}) and lower and upper 95% confidence intervals (LCI_{95} and $ULCI_{95}$) are calculated at the end of the exposure period. For the proposed listing threshold, a test product must demonstrate an acute LCI_{95} greater than 10 ppm in either fresh or salt water for all tested organisms, consistent with the acute toxicity thresholds proposed for dispersants. Using EPA toxicity classification scheme,[91] LC_{50} values ranging from 10 ppm to 100 ppm are classified as slightly toxic, and above 100 ppm substances are considered practically nontoxic to aquatic organisms. By proposing an LCI_{95} greater than 10 ppm threshold level the Agency is establishing an adequate safety margin without being overly restrictive.

(3) Listing of Non-Proprietary Nutrients

[91] USEPA 2010. http/www.epa.gov/oppefed1/ecorisk_ders/toera_analysis eco.htm#Ecotox.

The Agency recognizes there may be oil discharge situations where it is determined that the addition of mineral nutrients in the form of salts of nitrogen, phosphorus and potassium (i.e., fertilizers) to stimulate or enhance bioremediation may be the most effective and environmentally favorable mitigation method. However, commercially available formulations of mineral nutrients are not included on the Schedule. Currently, an OSC may only authorize the use of an agent not listed on the Schedule when its use is necessary to prevent or substantially reduce a hazard to human life. Further, RRTs are not currently able to preauthorize the use of mineral nutrients unless they are listed on the Schedule. To address these concerns, the Agency proposes to include on the Schedule a specific list of the following generic non-proprietary mineral nutrients: ammonium nitrate, ammonium phosphate, sodium nitrate, potassium nitrate, urea, sodium triphosphate, sodium phosphate, potassium phosphate, triple super phosphate, or any combination thereof. For these non-proprietary commercial nutrients, the Agency believes there is no need for submission of information that is readily available. Thus, the proposal requires no technical product data or further action on the part of a manufacturer prior to authorizing their use in response to a discharge event. However, this listing will be limited to products commonly formulated entirely of those nutrients listed. For products that may contain additional proprietary components or have unique proprietary formulations, the requirement for the submitter to provide the toxicity and efficacy data under the bioremediation agent category will apply. The Agency considered proposing a definition of "fertilizer" to specifically capture the listed bioremediation agents proposed to be exempted. However, fertilizer is a term commonly used to describe organic nutrients, which may contain substances that do not play a role in mitigating oil discharges, and that may have deleterious effects on the environment. Some non-proprietary fertilizers, for example, comprised of activated sewage sludge or manure, could contain unwanted pathogens and trace metals that could further stress the

environment if used in response to an oil discharge. Thus, EPA proposes to restrict the exemption under the bioremediation category only to those mineral nutrients listed above in order to avoid any confusion that could result from an exemption based on a fertilizer definition. The Agency requests comments on this approach.

e. **Solidifier Testing and Listing Requirements** §300.915(e)

The proposal defines solidifiers as polymeric substances that cause oil to become a coherent mass, preventing oil from leaching out or sinking into the water column, and that are collected and removed from the environment. For this agent category, the Agency proposes to revise the toxicity testing protocol and to establish a listing threshold for toxicity. Although these agents are intended to be removal from the environment, the proposed revisions and new toxicity listing threshold respond to concerns regarding the general increase in the use of chemical and biological agents as tools available for oil discharge responses.

(1) Solidifier Efficacy

There are currently no efficacy testing requirements in the NCP Subpart J for solidifiers to determine listing eligibility on the Schedule. These agents would have been captured by the current MOSCA agent category, which has no efficacy testing requirements, and which the Agency is proposing to eliminate. While the Agency is aware of existing protocols to determine the effectiveness of solidifiers, its focus has been on reviewing the protocols for dispersants and bioremediation agents, given that their mechanism of action allows them to be left in the environment, whereas solidifier are intended for removal from the environment. Therefore, the Agency at this time is not proposing an efficacy testing methodology for solidifiers, but is requesting comment on available methodologies it may be able to incorporate as the efficacy testing protocol for these agents.

(2) Solidifier Toxicity

Current Requirements: The current NCP Subpart J requires solidifiers to be tested for saltwater species toxicity. Solidifier toxicity is tested following the dispersant toxicity testing protocols in Appendix C to part 300 of the NCP.

Proposed Revisions: The proposal revises the toxicity testing requirements for solidifiers, including the testing protocol. While the Agency currently provides the toxicity testing results to the OSC to assist in authorization of use determinations, it is proposing to use these results to determine listing eligibility on the Schedule.

The proposed revisions to the testing protocols for solidifiers are detailed in Appendix C to part 300. The standard acute toxicity test protocol for solidifiers is based on EPA's protocol, *Methods for Measuring the Acute Toxicity of Effluents and Receiving Waters for Freshwater and Marine Organisms.* The Agency proposes to require solidifiers be tested for acute toxicity for the product alone using fresh water species *Ceriodaphnia dubia* and *Pimephales promelas*, or saltwater species *Americamysis bahia* and *Menidia beryllina*, or both, depending on the intended product use. The concentration of test product causing 50% lethality to the test organisms (LC_{50}) and lower and upper 95% confidence intervals (LCI_{95} and $ULCI_{95}$) are calculated at the end of the exposure period. For the proposed listing threshold, a test product must demonstrate an acute LCI_{95} greater than 10 ppm in either fresh or salt water, consistent with the acute toxicity thresholds proposed for dispersants. Using the EPA toxicity classification scheme,[92] LC_{50} values ranging from 10 ppm to 100 ppm are classified as slightly toxic and above 100 ppm substances are considered practically nontoxic to aquatic organisms. By proposing an LCI_{95} greater than 10 ppm threshold level the Agency is establishing an adequate safety margin without being overly restrictive.

[92] USEPA 2010. http/www.epa.gov/oppefed1/ecorisk_ders/toera_analysis eco.htm#Ecotox.

(3) Limitations

The Agency is proposing a conditional use listing for solidifiers. The proposal specifies that the listing of these products is limited to use only in those waters for which the product was tested and for which it met the listing threshold criteria. The Agency recognizes that products may yield effective results in certain environments and not in others. Products that may be effective in freshwater may not necessarily be so in saltwater, and vice versa. The Agency is proposing this limitation as a flexible alternative to product manufacturers, who may choose to test a product for a selected environment. The product will be listed and may only be authorized for use in those water environments for which it was tested and for which it met both the efficacy and toxicity listing criteria.

f. **Herding Agent Testing and Listing Requirements** §300.915(f)

The proposal defines herding agents as substances that form a surface film to control the thickness layer of oil and that are used to move oil horizontally across the water's surface. For this agent category, the Agency proposes to revise the toxicity testing protocol and to establish a listing threshold for toxicity. While these agents are intended to be removed from the environment, the proposed revisions and new toxicity listing threshold respond to concerns regarding the general increase in the use of chemical and biological agents as tools available for oil discharge responses.

Currently there is a test requirement for distinguishing surface collecting agents from other chemical agents (§300.915(c)(9), *Test to Distinguish Between Surface Collecting Agents and Other Chemical Agents*). Because the proposal eliminates surface collecting agents as a category and redefines herding agents to better reflect its mechanism of action, and given that the agents will need to be identified in order for the required testing to be submitted, the Agency is eliminating this requirement.

(1) Herding Agent Efficacy

There are currently no efficacy testing requirements in the NCP Subpart J for herding agents to determine listing eligibility on the Schedule. These agents would have been captured by the current surface collecting agent category, which has no efficacy testing requirements, and which the proposal eliminates. The Agency at this time is not proposing an efficacy testing methodology for herding agents, but is requesting comment on available methodologies it may be able to incorporate as the efficacy testing protocol for these agents.

(2) Herding Agent Toxicity

Current Requirements: The current NCP Subpart J requires herding agents to be tested for saltwater species toxicity. Herding agent toxicity is tested following the dispersant toxicity testing protocols in Appendix C to part 300 of the NCP.

Proposed Revisions: The proposal revises the toxicity testing requirements for herding agents, including the testing protocol. While the Agency currently provides the toxicity testing results to the OSC to assist in authorization of use determinations, we are proposing to use these results to determine listing eligibility on the Schedule.

The proposed revisions to the testing protocols for herding agents follow the same procedures as described for surface washing agents and are detailed in Appendix C to part 300. The standard acute toxicity test protocol for herding agents is based on EPA's protocol, *Methods for Measuring the Acute Toxicity of Effluents and Receiving Waters for Freshwater and Marine Organisms.* The Agency proposes to require herding agents to be tested for acute toxicity for the product alone using fresh water species *Ceriodaphnia dubia* and *Pimephales promelas*, or saltwater species *Americamysis bahia* and *Menidia beryllina*, or both, depending on the intended product use. The concentration of test product causing 50% lethality to the test organisms (LC_{50}) and lower and

upper 95% confidence intervals (LCI_{95} and $ULCI_{95}$) are calculated at the end of the exposure period. For the proposed listing threshold, a test product must demonstrate an acute LCI_{95} greater than 10 ppm in either fresh or salt water, consistent with the acute toxicity thresholds proposed for dispersants. Using the EPA toxicity classification scheme,[93] LC_{50} values ranging from 10 ppm to 100 ppm are classified as slightly toxic and above 100 ppm substances are considered practically nontoxic to aquatic organisms. By proposing an LCI_{95} greater than 10 ppm threshold level, the Agency is establishing an adequate safety margin without being overly restrictive.

The Agency is requesting comment on the procedure for preparing the stock solution for herding agents for the acute toxicity tests. Herding agents form a monolayer of surfactants on the water surface that result in the contraction of the oil slick.[94, 95] Herding agent composition may include hydrocarbons, fluorosurfactants and/or silicone-based surfactants, which suggests that a stock solution prepared using a WAF procedure similar to solidifiers may be appropriate. Any alternative procedure for preparing the stock solution must include an appropriate rationale in order for the Agency to be able to consider it for final action.

(3) Limitations

The Agency is proposing a conditional use listing for herding agents. The proposal specifies that these products may only be used in those waters for which the product was tested and for which it met the listing threshold criteria. The Agency recognizes that products may yield effective results in certain environments and not in others. Products that may be effective in freshwater may not necessarily be so in saltwater, and vice versa. The Agency is proposing this

[93] USEPA 2010. http/www.epa.gov/oppefed1/ecorisk_ders/toera_analysis eco.htm#Ecotox.
[94] Buist, I., Potter, S., Nedwed, T., (2011) Herding Agents to Thicken Oil Spills in Drift Ice for In Situ Burning: New Developments, IOSC, http://www.iosc.org/papers_posters/IOSC-2011-230-file001.pdf
[95] Buist, I., Nedwed, T., (2011) Using Herders for Rapid *In Situ* Burning Of Oil Spills on Open Water, IOSC, http://www.iosc.org/papers_posters/IOSC-2011-231-file001.pdf

limitation as a flexible alternative to product manufacturers, who may choose to test a product for a selected environment. The product will be listed and may only be authorized for use in those water environments for which it was tested and for which it met both the efficacy and toxicity listing criteria.

g. **Sorbent Requirements** §300.915(g)

The proposal defines sorbents as inert, insoluble substances that readily absorb and/or adsorb oil or hazardous substances. The proposed definition states that sorbents are generally collected and removed from the environment and may be used in their natural bulk form, or as manufactured products in particulate form, sheets, rolls, pillows, or booms. The list of sorbent materials in the proposed definition includes: natural organic substances (i.e., peat moss, straw, cellulose fibers, cork, corn cobs, feathers); inorganic/mineral compounds (i.e., volcanic ash, perlite, vermiculite, zeolite, clay); and synthetic compounds (i.e., polypropylene, polyethylene, polyurethane, polyester). The Agency proposes to identify a list of known sorbent materials as generic products, not on the Schedule, but as a separate publicly available list so that emergency responders can know and make use of such sorbents on oil discharges. The Agency is also proposing a process for submitters to request to include other products as sorbents if they can certify they meet the inert, insoluble criteria.

Current Requirements: Sorbents are currently not listed on the Schedule, but rather a list characterizing these materials is included in section 300.915(g). The current rule allows the OSC to request written certification from the manufacturers that their particular sorbent product is comprised solely of those materials identified in the rule. When a sorbent consists of materials that are not specifically listed in the current rule, the Agency issues written certifications, and sometimes requires testing and evaluation for possible listing on the Schedule.

Proposed Revisions: The Agency proposes to provide a Sorbent Product List that makes publicly available known non-proprietary sorbent materials. The Agency also proposes to allow submitters to request that other products be included in this list as sorbents if they can certify they meet the inert, insoluble criteria. The Agency recognizes that a sorbent material may consist of one or more substances not specifically identified in the proposed non-proprietary listing. For products consisting of one or more substances not specifically identified in the proposed generic listing, the Agency proposes submission requirements that include general and product characterization information required under paragraphs (a)(1) through (8), and paragraph (a)(13) of the proposed §300.915. These include name and contact information, identification as manufacturer or designated agent, product name/ brand/trademark(s), contact information for primary distributors or importers, product MSDS, conditions for product storage, product shelf life, and product label samples. The proposal would additionally require the certification statements required under paragraphs (a)(14)(D), (a)(15), and a(16) of the proposed §300.915. The submitter would be required to certify that the product does not contain specified bacterial, fungal, or viral pathogens or opportunistic pathogens above National Ambient Water Quality Criteria lowest density values; that the product does not contain specified contaminants above National Water Quality Standards levels; and that the product does not contain any of the prohibited agents. Finally, the submitter would be required to include a statement certifying the sorbent product is inert and insoluble and to provide information to support that claim such as data on the relative solubility and non-reactivity in fresh and/or salt water. The Agency will review the submission and make a determination to include the product as a sorbent on the Sorbent Product List. The Agency requests comments on this approach.

Testing Options:

The current Subpart J has no toxicity or efficacy testing requirements for sorbents. There are two ASTM standards for performance testing that could be applicable to sorbents: the ASTM F716 - 09 *Standard Test Methods for Sorbent Performance of Absorbents*, and the ASTM F726 - 06 *Standard Test Method for Sorbent Performance of Adsorbents*. These methods include laboratory tests that describe the performance of these products in removing oils that are not emulsified and other floating, immiscible liquids from the surface of water. While these methods are to be individually used as a basis for comparison of absorbents or adsorbents in a consistent manner, they are not useful for a comparison of absorbents with adsorbents, even though all absorbents exhibit adsorbent properties. Further, these agents are comprised of inert and insoluble materials that are generally removed from the environment after use, minimizing their potential harm. Thus, the Agency is not proposing to include efficacy or toxicity testing requirements for these agents and is requesting comments on this approach.

5. Submission of Confidential Business Information (CBI) – 300.950

Currently, manufacturers may assert a claim of confidential business information (CBI) for any information in their product package submissions to EPA. Typically, manufacturers claim as CBI the chemical identity (e.g. chemical name and chemical abstracts number [CAS]) and concentration (weight percent) of each chemical component in the product along with information about the formula or recipe for the product. The manufacturer may also claim certain other information in the technical product data, including technical data for sorbents, as CBI. EPA safeguards CBI information under the requirements in 40 CFR Part 2, subpart B.

Due to the unprecedented amount of dispersant used during the Deepwater Horizon incident, and the need for the public's right-to-know about chemicals intentionally discharged into the environment, EPA wanted to make public information about the chemicals in the dispersant

used, the results of air and water monitoring for these chemicals, and the concerns for human and environmental impact. A number of stakeholders wanted to know the chemical makeup of not only the dispersant used during the incident, but also that of all other dispersants on the Schedule. This resulted in both a number of Freedom of Information Act (FOIA) requests and a Notice of Intent to Sue for Failure to Perform Nondiscretionary Duties under the CWA, requesting that the Agency release CBI for all dispersants on the Schedule.

EPA worked with the manufacturer for the dispersant used on the Gulf in response to the Deepwater Horizon incident to make the product chemical information publicly available (see http://www.epa.gov/bpspill/). The Agency also worked with all other dispersant manufacturers to compile a comprehensive aggregate list of the chemicals used in all listed dispersants. During this process, some manufacturers recognized the need to make information available to the public, but also argued the need to protect proprietary business information from competitors. Companies provided information on the magnitude of resources expended to develop a product and test it for listing on the Schedule; disclosing the chemical components and/or formulae for their product, they argued, would allow domestic and international competitors to freely duplicate or reverse engineer their product and potentially drive them out of the market. EPA is aware that over 90% of businesses on the current Schedule are small, and that for some companies the product they developed for the Schedule is their only business. While companies could protect their product via the U.S. patent process, they would be required to release components and formulae information, which would be made public. Manufacturers voiced concern that not only others may use that information to tailor competitive products, but that the U.S. patent process would offer no protection against international competitors.

EPA believes that when chemical agents are used on oil discharges, it is critically important

for the public and all other stakeholders to have information regarding the chemicals being added to the environment, along with information about their toxicity and fate. This is particularly true for major discharge events where larger quantities of chemical or biological agents may be authorized for use. Prompt and accurate information will allow the public to evaluate and understand the potential human and environmental effects of these chemical agents. However, EPA is also sensitive to the need for companies to protect their investment in developing unique products. EPA agrees that the identity of product components and their concentrations are CBI and should remain confidential, and is not considering any proposals to make that information public. However, in balancing the need for public information with proprietary business needs, the Agency is proposing limitations to what submitters are allowed to claim as CBI. The proposal allows CBI to be claimed only for identity of product components and their concentrations (including all chemical components, microbiological cultures, enzymes, or nutrients); all other information submitted to EPA for listing a product on the Schedule will be made public. While providing confidential treatment for the identity of product components and their concentrations, the proposal allows public access to relevant health and environmental effects information. The Agency requests comments on whether this approach safeguards against duplication or reverse engineering of products by competitors.

Alternatively, the Agency considered maintaining the current approach of allowing CBI claims for any information in their submission. When the incident arises where affected stakeholders or the public wants access to specific information, the Agency would request the company substantiate its claims and make a determination whether to honor the claim or release the information to the public as provided in 40 CFR part 2, subpart B. The Agency also considered modifying the current approach by making the component identity and concentration information

public without further notice or action for major discharge events or SONS, or for events where a given amount of chemical or biological agent would be allowed for use. Finally, an option allowing for manufacturers to waive CBI claims only for certain chemical components for monitoring purposes (e.g. manufacturer identifies a "marker" component as a condition for listing) was also considered. The Agency does not believe continuing with the current approach, including the modifications considered, offers the appropriate balancing between public interest/business needs and is thus rejecting it.

EPA also considered developing aggregate list of components used in categories of chemical and biological agents for public disclosure. For example, a list of all the chemicals used in listed dispersants, a separate list for those substances used in surface washing agents, etc. This would allow information to be disassociated with specific products and protected from reverse engineering or duplication of products by competitors, while providing public access. The concern is for how to update the aggregate listing for new products without potentially revealing the components added to the list for that new product. Further, in the event of a major discharge or SONS, the interest will be for information and monitoring data specific to the product being used. Consequently, EPA is not adopting this approach but requests comments on these and other options to handle CBI while balancing all interests.

6. Addition of a Product to the Schedule - (300.955)

The proposal establishes the requirements for submitters to request a product be listed on the Schedule. It provides administrative information, such as the address where to submit the package, as well as details of the requirements for a complete submission package. Additionally, it addresses how a submitter may request a listing determination review and the requirements when there are changes in a listed product. Finally, the proposal addresses the process the Agency will follow to

review all new submissions, requests for review of decisions, and product changes, as well as how it will transition from the current Schedule to a new one that reflects the new and amended testing and data requirement.

Submission. The proposal updates the address in the current regulation.

Package contents. The proposal specifies a complete package must include, as follows:

- A company letter certifying all testing was conducted on representative product samples at a nationally or internationally accredited laboratory, that it was conducted in accordance with all technical rule requirements, and that all test results and product technical data and information reported are true and accurate;
- A numbered Table of Contents showing all required information and data submitted;
- All required data and information (both general and product category specific) in the order the requirements appear in the rule; and
- A separate inner envelope labeled: "CBI - TO BE OPENED BY THE PRODUCT SCHEDULE MANAGER ONLY", including the address for the submission package, if applicable.

Because of their intended function in responding to oil discharges, products listed on the Schedule will certainly impact the environment. It is important that the information provided by the submitter is true and accurate, as it serves as the basis for evaluating those potential environmental impacts. The Agency believes that it is appropriate for the submitter to be held accountable for the technical data and information provided to make these listing determinations. Thus, the proposal requires a certification of this information from the submitter.

The requirements for a table of contents and for the information to be organized as it

appears in the regulation are intended to make the Agency review process more efficient. These requirements will assist the Agency in conducting a quick and accurate review process both during the transition period, as well as for future submissions by generally simplifying the review process.

While the Agency needs to process packages containing information claimed to be CBI with additional safeguards, it is the responsibility of the submitter to ensure that this information comes to the Agency clearly identified as such. Therefore, the proposal requires a separate and clearly marked envelope for CBI to ensure proper handling.

EPA Review. The proposal maintains most of the existing Agency process for reviewing product submissions. A revision to the current process increases the number of days allowed for the Agency to complete its product review from the current 60 days to 90 days from the date of receipt. This proposed change considers the additional technical data and information proposed to be required, as well as the Agency's past experience with submission packages based on the current requirements.

EPA will first review the package for completeness and compliance with all data and information requirements and will contact the submitter to verify information, or to request clarification or additional information as necessary. Within the 90-day timeframe, the Agency will notify the submitter, in writing, of its decision to either list the product on the Schedule and under which category or categories, or of its decision to reject the submission as incomplete or non-compliant. The Agency will provide the submitters with its supporting rationale for any decision to reject the product submissions for listing. Submitters may revise incomplete submission packages to correct test results, data, or information deficiencies and resubmit them. Because the Agency will need a complete set of data and technical information to make a listing determination, the 90-day review time period will start anew once a complete package is resubmitted.

Request for review of decision. The proposal is not substantively changing the process for a submitter to request that the Agency review its determination on a product. If the Agency rejects a product for listing on the Schedule, the proposal continues to allow for a submitter to appeal to the EPA Administrator for a review of its determination. The proposal maintains the requirement that such a request be in writing, within 30 days of receipt of the written notification of EPA's decision not to list the product on the Schedule. The request to review the Agency's determination must include a clear and concise statement with supporting facts and technical analysis that demonstrates why the submitter believes the Agency's assessment of the product was incorrect. The proposal allows for the Administrator to request additional information or a meeting opportunity. Within 60 days of receipt of any such request, or within 60 days of receipt of any requested additional information, the proposal requires the Administrator or her designee to notify the submitter in writing of the review decision, maintaining the current timeframe.

Changes to a listed product. The proposal revises the provisions for notification of changes to a listed product. Submitters must continue to notify EPA in writing of any product modifications and any changes to the general product information submitted for listing on the Schedule. However, the proposal revises the notification requirement to include details of the specific changes, the reasons for such changes and the supporting data and information, while maintaining the provision allowing EPA to request additional information and clarification regarding these changes. For any changes to the chemical components and/or their concentrations, the proposed revisions would require retesting of the product according to the requirements for the product category, and the resubmission of a complete new package for a new review and listing determination by the Agency. While the Agency currently has and is retaining the option of requiring additional testing, it believes that when the chemical components or concentrations of a product change, an automatic

retesting requirement is merited. The Agency believes this requirement is appropriate when the identity of the product itself changes; the only way of evaluating the potential effects of these changes on the efficacy and toxicity of a new product formulation is to retest it. The Agency considered whether it was necessary to explicitly provide the flexibility to waive this requirement under extraordinary circumstances (i.e., a SONS event). However, OSCs already have broad authority to use agents in areas impacted or threatened by a release or discharge, whether the agent is identified or not on the Schedule, to prevent or substantially reduce an immediate threat to human life. In addition, the Agency considered whether there is a chemical concentration threshold that could accommodate minor adjustments to a product. For example, a producer may make a slight variation in a product formula to account for a food grade or technical grade chemical component. Such a threshold might be that if concentration changes vary by no more than 1%, no retesting is necessary. However, the Agency has no basis for such a threshold and requests comment on this approach with appropriate technical details. Because of this, and the concern for the potential impact reformulated products may have on the environment, the Agency is proposing to require retesting whenever the chemical components or concentrations of a product change. The Agency is also requesting comments on whether it should require the retesting of all products that have a change in composition, even if the only change is to chemical components that may considered inert.

Transitioning Listed Products from the Current Schedule to the New Schedule. The Agency believes it important that products on the current Schedule continue to be available during the transition period to a new Schedule that reflects the amended requirements. During this transition period, all products on the current Schedule will remain conditionally listed and the Agency will rely on it for planning and response activities. Because of the proposed revisions to test protocols

and listing criteria, and because of the additional test requirements, the Agency proposes that all products currently on the Schedule be retested, and that the new data and information be submitted to the Agency for reevaluation of the current listings within 24 months of the effective date of a final rule. The Agency believes this provides adequate time for submitters to prepare and submit new packages to EPA and for the Agency to review and make decisions on these products. For a product to be transitioned from the current Schedule to the new Schedule, manufacturers would be required to submit a new, complete package according to the amended test and listing criteria, and EPA would need to make a favorable finding to list the product on the new Schedule, either as currently listed or with modifications. Products on the current Schedule for which a new submission is not received, or that upon review of their submissions do not meet the revised listing criteria, would be removed from the Schedule at the end of the 24-month transition period. While the Agency is reviewing the new submission packages, planning and response authorities are encouraged to consider those products that based on existing data would meet the revised listing criteria. While the existing data is limited and developed with different protocols, considering the new requirements in selecting products for planning and response activities would provide an increased level of environmental protection. The Agency notes, however, that this is not a requirement, and that any product on the current Schedule may be authorized for use following the current protocols, until the 24-month transition period is over. The Agency is requesting comments on the practicability of this process and whether the 24-month period allows sufficient time for retesting and review.

7. Mandatory Product Disclaimer (300.965)

The current requirements provide that the listing of a product on the Schedule does not constitute approval or endorsement of the product. To avoid possible misinterpretation or

misrepresentation, the Agency currently requires that any label, advertisement, or technical literature that refers to the placement of the product on the Schedule must either reproduce in its entirety EPA's written statement that it will add the product to the Schedule under the current §300.920(a)(2) or (b)(2), or include the disclaimer set forth in the current §300.920(e). It remains the Agency position that listing a product on the Schedule does not constitute approval or endorsement of that product, nor a recommendation of its use. The Agency continues to believe that it is important to avoid any possible misinterpretation or misrepresentation of this policy. Thus, the requirement for a disclaimer to be included on any label, advertisement, or technical literature for the product is maintained. The proposal removes, however, the EPA written statement option; the Agency believes that given the advances in information technology it will be able to update the Schedule list within a reasonable timeframe, and that the option of producing the EPA letter of notification for a product listing is no longer necessary. The disclaimer language is proposed to be modified as follows:

> *[PRODUCT NAME] is listed on the National Contingency Plan (NCP) Product Schedule. This listing does NOT mean that EPA approves, recommends, licenses, or certifies the use of [PRODUCT NAME] on an oil discharge. This listing means only that data have been submitted to EPA as required by Subpart J of the NCP. Only a Federal On-Scene Coordinator (OSC) may authorize use of this product according to the NCP.*

The proposed revisions are intended to clarify that the use of these products is conditional to OSC authorization following the requirements set forth under the NCP regulations. The disclaimer language must continue to be conspicuously displayed in its entirety, and must be fully reproduced on all product literatures, labels, and electronic media including website pages. As discussed in the next section, a product can be removed from the Schedule if the submitter does not comply with the

disclaimer requirements, or makes any improper attempt to demonstrate the approval or endorsement of a product. The Agency requests comments on the proposed revisions.

8. Removal of a Product from the Schedule (300.970)

Products that are not properly used in the field may cause harm to human health and the environment, and may constitute violations of the CWA, and other federal, state, or local laws. Misleading, inaccurate, or incorrect statements within a product submittal package or within language that refers to the listing of a product on the Schedule may result in their improper or incorrect use. Falsification of federal documents, unsupported toxicity or efficacy claims, submission of incorrect product composition or use information, or withholding technical product data are some examples of these acts. To minimize potential misuse of listed products, the Agency believes it is appropriate to further clarify the criteria for the removal of a product from the Schedule. The proposal specifically includes, but does not limit, as causes for removal from the Schedule: any fraud or misrepresentation of a product on a submission to EPA or to any person or private or public entity regarding the composition or use of the product to remove or control oil discharges, including on labels, advertisements, or technical literature; the alteration of chemical components, concentrations, or use conditions of the product without proper notification to EPA; or the failure to print the disclaimer on all labels, advertisements, or technical literature. It also establishes a process for removal if the Agency obtains evidence of cause for removal. EPA would notify the submitter in writing, at the address of record, of its reasons for removal of the product from the Schedule. The proposal would allow for an appeals process similar to the one set forth for listing determinations. Appeals must be received within 30 days of receipt of EPA's removal notification and must contain a clear and concise statement with supporting facts and technical analysis demonstrating why the submitter believes EPA's decision was incorrect. Written

notification from the Administrator will be sent to the submitter within 60 days of any appeal, or within 60 days of receipt of any requested additional information. However, if no appeal is received within the time frame listed above the product would be delisted without further notice. The Agency requests comments on the proposed clarification of criteria for removal of products from the Schedule, and on the associated appeals process.

VI. Summary of Proposed Rule Provisions

This section summarizes the proposed changes to the rule. Subpart J has been renumbered to include new, consolidated, and revised sections. Some of rule sections have been retained, deleted, or moved in their entirety. The Table below provides an overview of existing rule and proposed rule citations for a quick reference.

Section 300.900, General, paragraphs (a) and (c) would be revised to reflect new and amended regulatory definitions for product categories.

Section 300.905, NCP Product Schedule, would be removed.

Section 300.910, Authorization of Use, would be revised and new paragraphs added to clarify planning and preauthorization responsibilities.

- Paragraph (a) would be revised to clarify the requirements and add new preauthorization plan review, concurrence, and withdrawal procedures.
- Paragraph (b) would be revised to clarify the requirements for using a listed product not addressed by a preauthorization plan and add new parameters for use considerations.
- Paragraph (c) would be revised to clarify the requirements for authorizing the use of burning agents on a case-by-case basis and add new parameters for use considerations.
- Paragraph (d) would be revised to clarify the exception requirements and add specific time frames for notification of continued agent use.

- Paragraph (e) would be revised to expand the prohibition to include nonylphenol (NP) or nonylphenol ethoxylates (NPEs) as components.
- Paragraph (f) would be revised to add new regulatory requirements for agent storage and use. Existing paragraph (f) requirements would be moved to new paragraph (g), Supplemental Testing and Information.
- New paragraph (g), Supplemental Testing and Information, would revise the regulatory text to clarify the requirements for supplemental testing.
- New paragraph (h), Recovery of Oil and Agents from the Environment, would add regulatory requirements for recovery of oils and agents from the environment.
- New paragraph (i), Notification of Agent Use, would add regulatory requirements for notification of agent use on an oil discharge.

New section 300.913, Monitoring the Use of Certain Agents, would add regulatory requirements for monitoring prolonged surface and subsurface dispersant application.

Section 300.915, Data Requirements, would be revised to consolidate general submission requirements applicable to all product categories. The section would be restructured to include new testing and listing requirements for specific product categories.

- Paragraph (a), Dispersants, would be revised to consolidate general testing and listing requirements from existing paragraphs (a), (b), (d), and (f). The paragraph would include revisions and new requirements for the identification of and testing for all product categories designated for listing. Existing paragraph (a) requirements specific to dispersants would be moved to new section 300.915(b), Dispersant Testing and Listing Requirements. The paragraph would be revised to add new toxicity and effectiveness testing requirements,

limitations for use, and new criteria for listing a dispersant to the Schedule. Existing paragraph (b) would be moved to new paragraph (c), Surface Washing Agent Testing and Listing Requirements.

- Paragraph (b), Surface Washing Agents, would be moved to new paragraph (c), Surface Washing Agent Testing and Listing Requirements. The paragraph would be revised to add new toxicity and effectiveness testing requirements, limitations for use, and new criteria for listing a surface washing agent to the Schedule. Existing paragraph (c), Surface Collecting Agents, would be deleted.
- Paragraph (d), Bioremediation Agents, would be revised to add new toxicity and effectiveness testing requirements, limitations for use, and new criteria for listing a bioremediation agent to the Schedule. Existing paragraphs (d)(9) and (10) were moved to new paragraph (a), General Product Information.
- Paragraph (e) would be revised to add new regulatory requirements for submission and listing of a solidifier. Existing paragraph (e), Burning Agents, would be deleted.
- Paragraph (f) would be revised to add new toxicity testing requirements, limitations of use, and criteria for listing a herding agent on the Schedule. Existing paragraph (f), Miscellaneous Oil Spill Control Agents, would be deleted.
- Paragraph (g), Sorbents, would be revised to add new exceptions for listing a sorbent to the Schedule.

Section 300.920, Addition of Products to Schedule, would be moved to new section 300.955, Addition of a Product to the Schedule.

- Paragraph (a) would be revised to include submission instructions for all product categories.

Existing paragraphs (a)(1)-(3) regulatory text specific to dispersant applications would be moved to new sections 300.915(b) and 300.955(c) and (d).

- Paragraph (b) would be revised to add new regulatory text for preparation of complete submission packages. Existing paragraph (b) regulatory text would be moved to new sections 300.955(c) and (d).
- Paragraph (c) would be revised to add regulatory text for EPA's review of submission packages and decision criteria for listing. Existing paragraph (c) would be moved to new section 300.950, Submission of Confidential Business Information (CBI).
- Paragraph (d) would be revised to add regulatory text for requesting a listing decision review. Existing paragraph (d) would be moved to new section 300.955(e), Changes to a Listed Product.
- Paragraph (e) would be revised to add new regulatory text for notification of changes to a listed product. Existing paragraph (e) would be moved to new section 300.965, Mandatory Product Disclaimer.
- New paragraph (f) would add new regulatory requirements for transitioning products on the current Schedule to the new Schedule.

New section 300.950, Confidential Business Information (CBI), would revise and clarify the allowable CBI claims in a submission package.

New section 300.965, Mandatory Product Disclaimer, would clarify the regulatory text for including a disclaimer statement on all product labels and literature.

New section 300.970, Removal of a Product from the Schedule, would add basis for removal of products from the Schedule, EPA notification of decision, and appeals process.

40 CFR part 300 – National Oil and Hazardous Substances Pollution Contingency Plan

Subpart J - Distribution Table

Current Citations	Proposed Rule Citations
§300.900 General	§300.900 General
§300.900(a)	§300.900(a)
§300.900(b)	§300.900(b)
§300.900(c)	§300.900(c)
§300.905 NCP Product Schedule	Deleted
§300.910 Authorization of Use	§300.910 Authorization for Agent Use
§300.910(a)	§300.910(a) Use of Agents Identified on the Schedule on Oil Discharges Addressed by a Preauthorization Plan
§300.910(b)	§300.900(b) Use of Agents Identified on the Schedule on Oil Discharges Not Addressed by a Preauthorization Plan
§300.910(c)	§300.900(c) Burning Agents
§300.910(d)	§300.910(d) Exceptions
§300.910(e)	§300.910(e) Prohibited Agents
§300.910(f)	§300.910(g) Supplemental Testing and Information
	§300.910(f) Storage and Use of Agents
	§300.910(h) Recovery of Oil and Agents

	from the Environment
	§300.910(i) Notification of Agent Use
	§300.913 Monitoring the Use of Certain Agents
	§300.913(a) Monitoring Dispersant Use on Surface Water
	§300.913(b) Monitoring Dispersant Used on Subsurface Waters
	§300.913(c) Burning Agent Monitoring
§300.915 Data Requirements	§300.915 Data and Information Requirements for Product Schedule Listing
§300.915(a)(1)-(12) Dispersants	§300.915(a)(1)-(20) General Product Information §300.915(b) Dispersant Testing and Listing Requirements
§300.915(b) Surface Washing Agents	§300.915(a)(1)-(20) General Product Information §300.915(c) Surface Washing Agent Testing and Listing Requirements
§300.915(c) Surface Collecting Agents	Deleted
$300.915(d) Bioremediation Agents	§300.915(a)(1)-(20) General Product Information §300.915(d) Bioremediation Agents

§300.915(e) Burning Agents	Deleted
§300.915(f) Miscellaneous Oil Spill Control Agents	Deleted
§300.915(g) Sorbents	§300.915(g) Sorbent Listing Requirements
§300.915(h) Mixed Products	Deleted
	§300.915(e) Solidifier Testing and Listing Requirements
	§300.915(f) Herding Agent Testing and Listing Requirements
§300.920 Addition of Products to Schedule	§300.955 Addition of a Product to the Schedule
§300.920(a)(1) Dispersants	§300.955(a) Submission §300.915(b) Dispersant Testing and Listing Requirements
§300.920(a)(2)	§300.955(c) EPA Review
§300.920(a)(3)	§300.955(d) Request for Review of Decisions
	§300.955(b) Package Contents
§300.920(b)(1) Surface Washing Agents, Surface Collecting Agents, Bioremediation Agents, and Miscellaneous Oil Spill Control Agents	§300.955(a) Submission

§300.920(b)(2)	§300.955(c) EPA Review
§300.920(c)	§300.950 Submission of Confidential Business Information (CBI)
§300.920(d)	§300.955(e) Changes to a Listed Product
	§300.955(f) Transitioning Listed Products from the Current Schedule to the New Schedule
§300.920(e)	§300.965 Mandatory Product Disclaimer
	§300.970 Removal of a Product from the Schedule

VII. Statutory and Executive Order Reviews

A. Executive Order 12866: Regulatory Planning and Review and Executive Order 13563: Improving Regulation and Regulatory Review

Under Executive Order 12866 (58 FR 51735, October 4, 1993), this action is a "significant regulatory action." This action raises novel legal or policy issues arising out of legal mandates, the President's priorities, or the principles set forth in the Executive Order. Accordingly, EPA submitted this action to the Office of Management and Budget (OMB) for review under Executive Orders 12866 and 13563 (76 FR 3821, January 21, 2011) and any changes made in response to OMB recommendations have been documented in the docket for this action.

In addition, EPA prepared an analysis of the potential costs and benefits associated with this action. This analysis is contained in the *Regulatory Impact Analysis For Proposed Revisions To The National Oil And Hazardous Substances Pollution Contingency Plan Regulations*. A copy of

the analysis is available in the docket for this action and the analysis is briefly summarized here.

The Agency expects the proposed rule would not cause a significant economic impact on a substantial number of small businesses. The incremental costs are anticipated to range from approximately $360,000 to $504,000 annually at 3% and 7% ($2012) annualization rates respectively. The benefits are anticipated to range from approximately $2.7 million to $3.8 million annually at 3% and 7% ($2012) annualized. The net benefits are anticipated to range from approximately $2.4 million to $3.3 million annually at 3% and 7% ($2012) annualization rates respectively. The monetized benefits are derived from having more effective products on the Schedule. The resulting compliance cost to the annual sales revenue ratio for the proposed rule for existing and new product manufacturers would be less than one percent in most instances.

B. Paperwork Reduction Act

The information collection requirements in this proposed rule have been submitted for approval to the Office of Management and Budget (OMB) under the *Paperwork Reduction Act*, 44 U.S.C. 3501 *et seq*. The Information Collection Request (ICR) document prepared by EPA has been assigned EPA ICR number EPA-HQ-OPA-2006-0090.

The ICR supporting today's proposed rule is largely self-implementing. The information collection is to ensure that: (1) the Agency has the necessary information to make Schedule listing determinations specific to the different product categories; (2) product use by owners or operators of facilities or vessels, or response personnel, in response to oil discharges is performed in accordance with all applicable requirements; and (3) the Agency can verify compliance as needed. Section 300.950 of the NCP contains provisions for confidentiality.

EPA has carefully considered the burden imposed upon the regulated community by the proposed regulations. EPA believes that the activities required are necessary and, to the extent

possible, has attempted to minimize the burden imposed. The minimum requirements specified in proposed today are intended to ensure that, when needed, products are used properly in the field to respond to an oil discharge in a manner protective of human health and the environment.

For the proposed requirements applicable to manufacturers, the burden to respondents over the three-year period covered by this ICR is estimated to be approximately 295 hours and $90,300 annually. The burden associated with reading and understanding the revisions to Subpart J of the NCP is estimated to require an estimated 5 hours per manufacturer each year. Preparing and submitting materials for listing a new product or reformulated product on the NCP Product Schedule is estimated to have a range of public reporting burden of approximately 18 to 34 hours per product, depending on the type of product listed. Preparing and submitting materials associated with re-testing an existing or new product on the NCP Product Schedule is estimated to have a public reporting burden of approximately 5 to 18 hours per product, depending on the type of product listed. Requesting a review of EPA's decision to not list a product on the Schedule is estimated to have a public reporting burden of approximately 10 hours per product. Written substantiation for sorbents is estimated to have a public reporting burden of 3 hours per product. There is no recordkeeping burden associated with listing a product on the Schedule. Burden is defined at 5 CFR 1320.3(b).

An agency may not conduct or sponsor, and a person is not required to respond to, a collection of information unless it displays a currently valid OMB control number. The OMB control numbers for EPA's regulations in 40 CFR are listed in 40 CFR Part 9.

To comment on the Agency's need for this information, the accuracy of the provided burden estimates, and any suggested methods for minimizing respondent burden, EPA has established a public docket for this proposed rule, which includes this ICR, under Document ID number EPA-HQ-

OPA 2006-0090. Submit any comments related to the ICR for the proposed rule to EPA and OMB; see **ADDRESSES** section at the beginning of this notice for where to submit comments to EPA. Send comments to OMB at the Office of Information and Regulatory Affairs, Office of Management and Budget, 725 17th Street, NW. Washington, DC 20503, Attention: Desk Officer for EPA. Since OMB is required to make a decision concerning the ICR between 30 and 60 days after [INSERT DATE OF PUBLICATION IN THE FEDERAL REGISTER], a comment to OMB is best assured of having its full effect if OMB receives it by [INSERT DATE 30 DAYS AFTER PUBLICATION IN THE FEDERAL REGISTER]. The final rule will respond to any OMB or public comments on the information collection requirements contained in this proposal.

C. Regulatory Flexibility Act (RFA)

The Regulatory Flexibility Act (RFA) generally requires an agency to prepare a regulatory flexibility analysis of any rule subject to notice and comment rulemaking requirements under the Administrative Procedure Act or any other statute unless the agency certifies that the rule will not have a significant economic impact on a substantial number of small entities. Small entities include small businesses, small organizations, and small governmental jurisdictions.

For purposes of assessing the impacts of today's proposed rule on small entities, small entity is defined as: (1) as defined by the Small Business Administration's (SBA) regulations at 13 CFR 121.201; (2) a small governmental jurisdiction that is a government of a city, county, town, school district or special district with a population of less than 50,000; or (3) a small organization that is any not-for-profit enterprise that is independently owned and operated and is not dominant in its field.

After considering the economic impacts of today's proposed rule on small entities, I certify that this action will not have a significant economic impact on a substantial number of small entities. The small entities directly regulated by this proposed rule are product manufacturers and laboratories

and state and local governments that are involved in product development, testing and use for oil discharge response. We have determined that this proposed rule does not have a significant impact on a substantial number of small entities.

Nonetheless, EPA has tried to reduce the impact of this rule on small entities in developing the regulatory requirements that balance the costs and burden while addressing the environmental protection concerns. We continue to be interested in the potential impacts of this proposed rule on small business entities and welcome comments on the issues related to such impacts.

D. Unfunded Mandates Reform Act

This action contains no Federal mandates under the provisions of Title II of the Unfunded Mandates Reform Act of 1995 (UMRA), 2 U.S.C. 1531-1538 for State, local, or tribal governments or the private sector. This proposed rule imposes no new enforceable duty on any state, local or tribal governments or the private sector. UMRA excludes from the definition of "Federal intergovernmental mandate" and "Federal private sector mandate" duties that arise from conditions of Federal assistance. UMRA generally excludes from the definition of "federal intergovernmental mandate" duties that arise from participation in a voluntary Federal program. UMRA also excludes from the definition of "Federal private sector mandate" duties that arise from participation in a voluntary Federal program. Therefore, we have determined that this proposed rule is not subject to the requirements of sections 202 or 205 of UMRA. This action is also not subject to the requirements of section 203 of UMRA because it contains no regulatory requirements that might significantly or uniquely affect small governments.

E. Executive Order 13132: Federalism

This action does not have federalism implications. It will not have substantial direct effects on the States, on the relationship between the national government and the States, or on the distribution of

power and responsibilities among the various levels of government, as specified in Executive Order 13132. This proposal does not alter the parameter already defined in the NCP of how state, local, and federal agencies cooperate in responding to oil spills and how to consult with the OSC and RRT when considering the use of products on the Schedule. Thus, Executive Order 13132 does not apply to this action. In the spirit of Executive Order 13132, and consistent with EPA policy to promote communications between EPA and State and local governments, EPA specifically solicits comment on this proposed action from State and local officials.

F. Executive Order 13175: Consultation and Coordination with Indian Tribal Governments

Subject to the Executive Order 13175 (65 FR 67249, November 9, 2000) EPA may not issue a regulation that has tribal implications, that imposes substantial direct compliance costs, and that is not required by statute, unless the Federal government provides the funds necessary to pay the direct compliance costs incurred by tribal governments, or EPA consults with tribal officials early in the process of developing the proposed regulation and develops a tribal summary impact statement.

EPA has concluded that this action may have tribal implications. However, it will neither impose substantial direct compliance costs on tribal governments, nor preempt Tribal law, similarly to the effect on states. EPA will be consulting with tribal officials early in the process of developing this regulation to permit them to have meaningful and timely input into its development. In the spirit of Executive Order 13175 and consistent with EPA policy to promote communications between EPA and tribal governments, EPA specifically solicits comment on this proposed rule from tribal officials.

G. Executive Order 13045: Protection of Children from Environmental Health Risks and Safety Risks

EPA interprets EO 13045 (62 F.R. 19885, April 23, 1997) as applying only to those regulatory actions that concern health or safety risks, such that the analysis required under section 5-501 of the EO has the potential to influence the regulation. This action is not subject to EO 13045 because it does not establish an environmental standard intended to mitigate health or safety risks to children.

H. Executive Order 13211: Actions That Significantly Affect Energy Supply, Distribution or Use

This proposed rule is not a "significant energy action" as defined in Executive Order 13211, "Actions Concerning Regulations that Significantly Affect Energy Supply, Distribution, or Use" (66 FR 28355, May 22, 2001) because it is not likely to have a significant adverse effect on the supply, distribution, or use of energy. Today's proposed rule would not cause reductions in the supply or production of oil, fuel, coal, or electricity; nor would it result in increased energy prices, increased cost of energy distribution, or an increased dependence on foreign supplies of energy.

I. National Technology Transfer and Advancement Act

Section 12(d) of the National Technology Transfer and Advancement Act of 1995 ("NTTAA"), Public Law No. 104-113, 12(d) (15 U.S.C. 272 note) directs EPA to use voluntary consensus standards in its regulatory activities unless to do so would be inconsistent with applicable law or otherwise impractical. Voluntary consensus standards are technical standards (e.g., materials specifications, test methods, sampling procedures, and business practices) that are developed or adopted by voluntary consensus standards bodies. NTTAA directs EPA to provide Congress, through OMB, explanations when the Agency decides not to use available and applicable voluntary consensus standards.

The rulemaking involves technical standards. The Agency conducted a search to identify

potentially applicable voluntary consensus standards for efficacy testing. However, we identified no such standards. Therefore, EPA developed the Baffled Flask Effectiveness Test and the Bioremediation Efficacy Test required in Appendix C of this proposed rule. Voluntary consensus standards developed by ASTM are recommended for several product property data points, such as pH, flash point and pour point. The product toxicity testing relies on existing protocols that are universally accepted. EPA welcomes comments on this aspect of the proposed rulemaking and, specifically, invites the public to identify potentially-applicable voluntary consensus standards for product efficacy and to explain why such standards should be used in this regulation.

J. Executive Order 12898: Environmental Justice

Executive Order 12898, "Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations" (59 FR 7629 (February 11, 1994)) establishes federal executive policy on environmental justice. Its main provision directs federal agencies, to the greatest extent practicable and permitted by law, to make environmental justice part of their mission by identifying and addressing, as appropriate, disproportionately high and adverse human health or environmental effects of their programs, policies, and activities on minority populations and low-income populations in the United States.

EPA is committed to addressing environmental justice concerns and has assumed a leadership role in environmental justice initiatives to enhance environmental quality for all citizens of the United States. The Agency's goals are to ensure that no segment of the population, regardless of race, color, national origin, income, or net worth bears disproportionately high and adverse human health and environmental impacts as a result of EPA's policies, programs, and activities. In response to Executive Order 12898, EPA's Office of Solid Waste and Emergency Response (OSWER) formed an Environmental Justice Task Force to analyze the array of

environmental justice issues specific to waste programs and to develop an overall strategy to identify and address these issues (OSWER Directive No. 9200.3-17). To address this goal, EPA conducted a qualitative analysis of the environmental justice issues under this proposed rule.

Under the National Oil and Hazardous Substances Pollution Contingency Plan (NCP), Regional Response Teams (RRT) and Area Committees are required to address, as part of their planning activities, the desirability of using appropriate chemical or biological agents, or other spill mitigating devices. In addition, the OSC, under authority granted by the NCP, must respond to an oil spill in a diligent and effective manner to protect human health and the environment. If chemical or biological agents are needed, the OSC must coordinate with the RRT and Area Committees before their use is authorized. In both cases, the RRT, Area Committees and OSC will address a broad array of oil spill response and mitigation issues, including the potential for environmental justice concerns. Historically, EPA has not found any evidence that the use of chemical or biological agents on the Product Schedule on oil spills in the US has had any disproportionate effect on any EJ communities. However, EPA will monitor the implementation of the rule to ensure the planned or actual use of chemical or biological agents has no disproportionate effect on any EJ communities.

EPA has determined that this proposed rule will not have disproportionately high and adverse human health or environmental effects on minority or low-income populations because it increases the level of environmental protection for all affected populations without having any disproportionately high and adverse human health or environmental effects on any population, including any minority or low-income population. This proposed rule is consistent with EPA's Environmental Justice Strategy and the OSWER Environmental Justice Action Agenda.

List of Subjects in 40 CFR part 300

Environmental protection, Air pollution control, Chemicals, Hazardous materials, Hazardous Substances, Incorporation by reference, Intergovernmental relations, Oil spills, Dispersants, Bioremediation, Surface washing agents, Oil spill mitigating devices, Sorbents, Regional response teams, and Area contingency planning.

Dated: _____

_____

Lisa P. Jackson

Administrator

To: Matthiessen, Craig[Matthiessen.Craig@epa.gov]
Cc: Wilson, Gregory[Wilson.Gregory@epa.gov]
From: Principe, Vanessa
Sent: Mon 7/27/2015 6:33:43 PM
Subject: IPIECA Dispersants: Subsea Use Draft Guidance / in Preparation for IMO Subsea Guidance Meeting
IPIECA_Dispersants_Subsea Use _OEM_7_27_15.docx

FYI – Attached are our comments on the IPIECA draft guidance that the workgroup wanted to consider adopting as IMO Subsea Guidance.

We have overarching concerns with the guidance's approach to NEBA, and with some very general statements and scientific "conclusions" that we should be very careful agreeing to as part of a government product. We will be recommending that we do not adopt this document.

V

Proposed Changes to Subpart J of the National Contingency Plan

The U.S. Environmental Protection Agency (EPA) is proposing to amend Subpart J of the National Contingency Plan (NCP) requirements for the use of dispersants, and other chemical and biological agents to respond to oil spills in waters of the U.S. The public has the opportunity to comment on this proposed rule during a 60-day period following its publication in the *Federal Register.*

Background

In 1994, EPA revised the NCP in response to the passage of the Oil Pollution Act of 1990. The final rule established the current Subpart J regulatory requirements with respect to authorization of use, data requirements, dispersant effectiveness and toxicity testing protocols, surface washing agent testing protocol, bioremediation agent testing protocols, and requirements for adding products to the Schedule.

Why is EPA proposing these amendments?

In April 2010, the Deepwater Horizon underwater oil well blowout released significant quantities oil into the Gulf of Mexico. Dispersants were deployed over several miles of the Gulf both applied on surface slicks and directly injected into the oil gushing from the well riser, totaling close to two million gallons of dispersants. This unprecedented use of dispersants raised many questions about efficacy, toxicity, environmental impacts and monitoring challenges that EPA now seeks to address through these proposed revisions to Subpart J..

These proposed changes would help to ensure that chemical and biological agents have met rigorous efficacy and toxicity requirements, and that product manufacturers have provided important use and safety information. This would equip the planning and response community with the proper information to evaluate the use of products to effectively mitigate health and environmental effects from oil discharges from occurring in navigable waters and other areas of concern.

What changes are proposed?

The proposed rule would amend Subpart J requirements for the use of dispersants, and other chemical and biological agents to respond to oil spills and considers concerns that arose during the Deepwater Horizon response regarding toxicity, efficacy, long-term environmental impacts, endangered species protection, and human health. Additionally, revisions are proposed to area planning requirements for dispersant use authorization and to toxicity and efficacy testing methodologies and listing thresholds, and advanced monitoring techniques are proposed. The revisions are a major component of EPA's effort to inform the use of dispersants and other chemical or biological agents, to respond to oil spills based on the federal government's experiences in the Gulf. Specifically, the rule proposes changes to:

- *Definitions.* Revise definitions for Bioremediation agents, Burning agents, Chemical agents, Dispersants, Sinking agents, Sorbents, and Surface washing agents; new definitions for Bioaccumulation, Bioconcentration, Biodegradation, Biological agents, Herding agents, Products, and Solidifiers; and remove the definitions for Miscellaneous Oil

Spill Control Agent (MOSCA) and Surface collecting agents.

- *General Requirements.* Revise to reflect new and amended regulatory definitions.
- *Authorization of Use.* Revise to clarify planning and preauthorization responsibilities, establish limitations and prohibitions on the use of certain agents, establish requirements for storage and use of agents, clarify authorities for requiring supplemental testing and information on agents, establish requirements for agent recovery from the environment, and provide notification of agent use.
- *Monitoring Requirements for Agent Use.* New monitoring requirements for major and/or continuous discharges.
- *General Product Information for Schedule Listing.* Revise and add new requirements, including designation of and testing for all product categories under which the listing is requested, Material Safety Data Sheets, sample product labels, shelf life, collection and recovery, persistence in the environment, storage and use conditions, physical and chemical properties, component identities, concentration limits on National Water Quality Standards and Criteria contaminants, laboratory accreditations, submission of all testing data and calculations, production capabilities, and any other data or certification informing the product's performance capabilities or environmental benefits.
- *Dispersant Testing and Listing Requirements.* New efficacy testing methodology using a baffled-flask test, new developmental and sub-chronic toxicity testing requirements, revisions to acute toxicity testing methodologies, and revisions to the listing criteria, and new use limitations to saltwater environments.
- *Surface Washing Agent Testing and Listing Requirements.* Revise the acute toxicity testing methodology and listing requirements, new efficacy testing requirements and listing criteria, and new use limitations based on product testing for salt and/or freshwater environments.
- *Bioremediation Testing and Listing Requirements.* Revise the efficacy and acute toxicity testing methodologies and listing criteria, new exceptions for specified products, and new use limitations based on product testing for salt and/or freshwater environments.
- *Solidifier Testing and Listing Requirements.* Revise the acute toxicity testing methodology and listing criteria, and new use limitations based on product testing for salt and/or freshwater environments.
- *Herding Agent Testing and Listing Requirements.* Revise the acute toxicity testing methodology and listing criteria, and new use limitations based on product testing for salt and/or freshwater environments.
- *Sorbent Testing and Listing Requirements.* Publicly available list of known, non-proprietary sorbents to be made available in lieu of listing sorbents on the Schedule; requirements for data, information and certifications for sorbent products with components other than those specifically identified as non-proprietary.
- *Submissions of Confidential Business Information.* Revise the allowable confidential business information claims and reporting procedures.
- *Addition of a Product to the Schedule.* Revise submission requirements, including the package contents, EPA's review, request for review of decision, changes to a listed product, and transitioning of listed products from the current Schedule to the new Schedule.
- *Mandatory Product Disclaimer.* Revise the product disclaimer requirements.

- *Removal of a Product from the Schedule.* Revise basis for removal and appeals process.
- *Appendix E to Part 300 – Oil Spill Response.* Remove this appendix to the NCP.

How can you comment on the proposal?

Comments must be received on or before [Insert date 60 days after publication in the Federal Register]. Comments must be identified by Docket ID No. EPA-HQ-OPA-2006-0090, and submitted by one of the following methods:

- Federal Rulemaking Portal: http://www.regulations.gov. Follow the on-line instructions for submitting comments.
- Mail: The mailing address of the docket for this rulemaking is EPA Docket Center (EPA/DC), Docket ID No. EPA-HQ-OPA-2006-0090, 1200 Pennsylvania Avenue, NW., Washington, DC 20460.
- Hand Delivery: Such deliveries are only accepted during the Docket's normal hours of operation, and special arrangements should be made for deliveries of boxed information. The Public Reading Room is open from 8:30 a.m. to 4:30 p.m., Monday through Friday, excluding legal holidays. The telephone number for the Public Reading Room is 202-566-1744.

For More Information

Read the Proposed Subpart J Rule Amendment:
http://www.epa.gov/emergencies/

Visit the EPA Office of Emergency Management Web site:
http://www.epa.gov/emergencies

Call the National Contingency Plan Product Schedule Information Line
(202) 260-2342 (phone)

To Report an Oil or Chemical Spill

Call the National Response Center:
(800) 424-8802 or (202) 267-2675
TDD (202) 267-4477

EPA NEWS ADVISORY

Proposed Amendments to Subpart J Product Schedule of the National Oil and Hazardous Substances Pollution Contingency Plan

Contact: Stacy Kika

The U.S. Environmental Protection Agency (EPA) is proposing to amend requirements under Subpart J of the National Oil and Hazardous Substances Pollution Contingency Plan (NCP) to address concerns raised by the regulated community and response officials, and to incorporate scientific advances and lessons learned from oil discharges, particularly the recent Deepwater Horizon event.

Specifically, EPA is proposing to amend the Subpart J Product Schedule regulatory requirements for a submitter to have chemical, biological, or other spill mitigating substances listed on the Product Schedule, to revise the efficacy and toxicity testing protocols, to add new listing criteria, and to require additional health and safety information. Additionally, EPA is proposing amendments to the area planning requirements for dispersant use authorization, and new monitoring techniques for certain major discharges, including data reporting. EPA is also proposing new evaluation criteria for removing products from the Schedule. These changes would ensure that the emergency response community has the information it needs to make informed spill response decisions in the field to protect human health and the environment.

More information about the proposed NCP Subpart J Product Schedule Rule is available at http://www/epa/gov/emergencies.

To: Wilson, Gregory[Wilson.Gregory@epa.gov]
From: Principe, Vanessa
Sent: Thur 6/4/2015 3:23:48 PM
Subject: FW: R&T Plan Ch 7&8
Ch 78 Member Draft 2015-05-13_OEM_5_27_15.docx

-----Original Message-----
From: Vocke, William CIV [mailto:William.T.Vocke@uscg.mil]
Sent: Thursday, June 04, 2015 11:10 AM
To: Principe, Vanessa
Subject: R&T Plan Ch 7&8

Hi Vanessa,

FYI - Attached are the changes made to Chapters 7&8 based on your comments.

Bill

To: Semeniuk, Michael[Semeniuk.Michael@epa.gov]
Cc: Matthiessen, Craig[Matthiessen.Craig@epa.gov]; Wilson, Gregory[Wilson.Gregory@epa.gov]
From: Principe, Vanessa
Sent: Mon 6/1/2015 3:32:09 PM
Subject: FW: API and NOIA Comments on EPA Subpart J Proposed Rule - Correction
removed.txt
API and NOIA Comments on EPA Subpart J Proposed Rule with Corrected PNG Attachment - Amended 05.29.15.pdf

Michael:

This is a follow up to my voice mail. For the request below, is there a protocol to be followed? The comment period for this docket closed on April 22, and it is my understanding that regulations.gov does not accept late comments. Does the Agency has set procedures to accept these and flag them as late? I do not want any official comment to the Agency be an email to my in-box.

Thru FDMS I would have access to upload a "Supporting Document." I do NOT want to classify this as such.

Please advise. Thanks, V

From: Emily Hague [mailto:Hague@api.org]
Sent: Friday, May 29, 2015 11:45 AM
To: Matthiessen, Craig; Wilson, Gregory; Principe, Vanessa
Cc: Raymond Bradley
Subject: API and NOIA Comments on EPA Subpart J Proposed Rule - Correction

Hello Craig, Greg and Vanessa –

API and NOIA submitted its comment letter on the EPA Subpart J Proposed Rule along with an independent economic assessment on April 22nd. It was subsequently discovered that, due to an error, the assessment undervalued existing stockpiles of dispersants in the U.S. inventory. We have corrected the error in the assessment and would like to resubmit our comment letter with

the corrected version of the assessment. No changes have been made to the comment letter itself.

Please consider the attached submission as our comments on the Proposed Rule.

Thank you,

Emily

Emily Kennedy Hague

202-682-8260

hague@api.org

This transmission contains information that is privileged and confidential and is intended solely for use of the individual(s) listed above. If you received the communication in error, please notify me immediately. Any dissemination or copying of this communication by anyone other than the individual(s) listed above is prohibited.

From: Emily Hague
Sent: Wednesday, April 22, 2015 6:08 PM
To: 'Matthiessen, Craig'; Wilson, Gregory (Wilson.Gregory@epa.gov); Principe, Vanessa (Principe.Vanessa@epa.gov)
Cc: Raymond Bradley
Subject: API and NOIA Comments on EPA Subpart J Proposed Rule

Hello Craig, Greg and Vanessa –

API and NOIA submitted the attached comments on the EPA Subpart J Proposed Rule this afternoon through Regulations.gov. Attached is our letter for your reference. We also plan to provide this letter by mail to Mathy Stanislaus.

Thank you for this opportunity to comment,

Emily

Emily Kennedy Hague

Senior Policy Advisor, Upstream & Industry Operations

American Petroleum Institute

1220 L Street, NW

Washington, DC 20005

Ph: 202-682-8260

Fx: 202-682-8426

hague@api.org

This transmission contains information that is privileged and confidential and is intended solely for use of the individual(s) listed above. If you received the communication in error, please notify me immediately. Any dissemination or copying of this communication by anyone other than the individual(s) listed above is prohibited.

May 29, 2015

U.S. Environmental Protection Agency
EPA Docket Center (EPA/DC)
Docket ID No. EPA–HQ–OPA–2006–0090
1200 Pennsylvania Avenue NW 20460

Submitted via electronic submission to http://www.regulations.gov **and by mail**

Dear Dr. Gregory Wilson,

On April 22, 2015, the American Petroleum Institute and the National Ocean Industries Association (collectively, "we") submitted a letter providing our comments on the U.S. Environmental Protection Agency's Proposed Rule to amend Subpart J of the National Oil and Hazardous Substances Contingency Plan. 80 Fed. Reg. 3380 (Jan. 22, 2015).

As an attachment to our April 22 letter, we provided an independent economic assessment of the Proposed Rule. It was subsequently discovered that, due to an error, the assessment undervalued existing stockpiles of dispersants in the U.S. inventory. Accordingly, we are now re-submitting our comment letter, attached to which is a corrected version of the assessment. There are no changes to the letter itself.

Please consider the attached submission as our comments on the Proposed Rule. We again thank you for your time and attention to our comments. Please do not hesitate to contact Emily Hague (202-682-8260, Hague@api.org) or Nicolette Nye (202-465-8463, nnye@noia.org) if you have any questions.

Sincerely,

Erik Milito
Group Director, Upstream and
Industry Operations
American Petroleum Institute

Nicolette Nye
VP Communications & Industry Affairs
National Ocean Industries Association

ED_001324_00007732-00001

April 22, 2015

U.S. Environmental Protection Agency
EPA Docket Center (EPA/DC)
Docket ID No. EPA–HQ–OPA–2006–0090
1200 Pennsylvania Avenue NW 20460

Submitted via electronic submission to http://www.regulations.gov **and by mail**

Dear Dr. Gregory Wilson,

On January 22, 2015, the U.S. Environmental Protection Agency (EPA or the Agency) issued a Proposed Rule to amend Subpart J of the National Oil and Hazardous Substances Contingency Plan that governs the use of dispersants, other chemical and biological agents, and other spill mitigating substances when responding to oil discharges into the waters of the United States. 80 Fed. Reg. 3380 (January 22, 2015). Comments are due by April 22, 2015. The following comments to the docket are provided by the American Petroleum Institute (API) and the National Ocean Industries Association (NOIA) (collectively, "we") on behalf of our member companies based on review of the above-cited Proposed Rule.

API is a national trade association that represents over 625 members involved in all aspects of the oil and natural gas industry, including the exploration and production of both onshore and offshore Federal resources. The U.S. oil and natural gas industry supports 9.8 million U.S. jobs and 8 percent of the U.S. economy, delivering tens of millions per day in revenue to our government. API members provide most of the nation's energy and are backed by a growing grassroots movement of more than 25 million Americans.

NOIA, founded in 1972, represents more than 270 companies among all segments of the offshore industry with an interest in the exploration and production of both traditional and renewable energy resources on the nation's outer continental shelf (OCS). NOIA's mission is to secure reliable access and a fair regulatory and economic environment for the companies that develop the nation's valuable offshore energy resources in an environmentally responsible manner.

API, NOIA, and our respective members recognize the important mission of the EPA in protecting human health and the environment and share in this mission. More than most industries, the oil and natural gas industry understands the nature of risk and places at the center of its concern comprehensive planning for and effective management of risks in order to safeguard human well-being and protect environmental resources. To this end, woven into every operational endeavor are release prevention measures, preparedness methods and plans for rapid, effective response should an incident resulting in a release occur. Developed alongside regulatory agencies, operators submit to and follow a distinct set of response principles which protect shared values. The framework by which these principles are employed and tradeoffs are made in order to protect these shared values is called Net Environmental Benefit Analysis (NEBA). NEBA is a process used by the response community for making the best choices to minimize impacts of oil releases on people and the environment. NEBA aids in protecting people and the environment, and is the matrix through which every decision is made in a response event. In the offshore context, oil dispersing agents are a primary tool in the NEBA process and the response toolkit, as they

are a principal tool used for protecting human health and defending environmentally sensitive areas. When a release occurs, judgment and response speed are critical. In an offshore setting, dispersants are essential in creating a safe environment and are a key ingredient in marshaling an effective, swift response.

API, NOIA, and our respective members understand that through this Proposed Rule, EPA's goal is to develop a regulation which protects people and preserves natural resources. We too wish to accomplish the same goal, based upon sound scientific research and grounded in real-world, outside-the-laboratory scenarios and taking into account incident-specific risk. While the original intent of the dispersant rule was simply to provide for a schedule of dispersants and other chemicals that may be used in carrying out the National Contingency Plan (NCP), the Proposed Rule would essentially expand its purpose to an operational rulemaking which would significantly alter operational procedures.

In summary, the Proposed Rule, if implemented, would be substantially problematic for all stakeholders, including the response community, which is required to carry out an effective spill response. The Proposed Rule would also be problematic for the Federal and state ag ency community interests, as well as the public with regard to increasing environmental and community impacts. Although the Agency indicates that it seeks to "*encourage the development of safer and more effective spill mitigating products, and would bette r target the use of these products to reduce the risks to human health and the environment*" and "*ensure that On-Scene Coordinators (OSCs), Regional Response Teams (RRTs), and Area Committees have sufficient information to support agent preauthorization or authorization of use decisions*," API, NOIA, and our respective members submit that the Proposed Rule falls short of these objectives and in many cases would have the opposite effect . For example, and at the center of our concern, is the proposed dispersed oil toxicity test and its threshold which could result in the elimination of many dispersants and potential future dispersant s. E ven if the dispersants themselves were completely non-toxic, the proposed dispersed oil toxicity test and associated pass/fail threshold could prevent all dispersants from being listed and available for use in a spill response . The Proposed Rule covers activities broader than mere scheduling criteria for dispersants. The added complexity resulting from the Proposed Rule in auth orization, monitoring and product listing, in addition to unfounded and overly corrective assumptions about dispersant effects described in the Proposed Rule, will not assist the Unified Command during an oil spill . Instead, it will likely complicate and limit its ability to respond in the most effective manner to ensure the health and safety of people and minimize environmental and community impacts. Also, prohibiting the application of dispersants during an oil spill response or delaying their use will significantly increase the economic and environmental damage of oil spills.

API and NOIA encourage the Agency to review the comments contained herein , incorporate the corresponding recommendations in a supplemental proposed rule, and provide the public and other Federal government and state stakeholder agencies with an opportunity to provide another round of comments be fore implementing a final rule. We also recommend that EPA initiat e a Negotiated Rulemaking under the Negotiated Rulemaking Act of 1990 to establish a negotiating committee under the Federal Advisory Committee Act to develop draft regulatory text on a consensus basis for adoption and implementation as a proposed supplemental proposed rule by EPA. A balanced negotiating committee made up of response experts – from the Federal Agencies who oversee oil spill response and from industry and relevant third-party experts – would add value to further Proposed Rule revisions and ensure that Subpart J is aligned with the latest scientific consensus on, and avoid an artificial bias against, the use of dispersants. As an example, Negotiated Rulemaking was successfully used to implement vessel and facility response planning requirements under the Oil Pollution Act of 1990. At a minimum, we believe that E PA should facilitate a discussion between the response experts before finalizing the Proposed Rule.

We first provide general comments followed by detailed comments on a section -by-section basis for EPA's consideration below.

GENERAL COMMENTS

Industry Shares EPA's Goal Of Establishing Sound Dispersants Policy Grounded In Science and Executable in Real-World Spill Scenarios

In every operation, the goal of industry is to never have an oil spill, and it invests extensive time and effort to prevent spills from occurring. Industry and government have a shared interest in a unified, coordinated, and proactive approach that allows us to protect our common values. These values include the health and safety of our citizens, sensitive ecosystems, the local business community and economy, and the vitality and sustainability of tourism and other key community industries, such as fishing.

To achieve a well-managed response and the protection of our common values, the primary guiding principle is to stop the source of a spill as quickly as possible with the goals of ensuring the health and safety of people and minimizing environmental and community impacts. In many spill circumstances, dispersants play a critical role in achieving these goals, perhaps most importantly by reducing risks associated with Volatile Organic Compounds (VOCs) in the immediate vicinity of the spill that could be both a human health and safety issue, and should continue to be a response option promptly available to the Unified Command. The Proposed Rule could jeopardize the protection of the common values by potentially precluding even completely non-toxic dispersants from being listed due to the proposed dispersed oil toxicity tests and associated pass/fail thresholds. Scientific studies have shown that approximately 97.5% of the TPH constituents in dispersed oil toxicity tests come from the oil (not the dispersant), so the proposed laboratory tests will measure the toxicity of oil (not the dispersant).[1] If the standardized oils (i.e., ANS and IFO-120) required to be used in the proposed test have toxicity values (LC50) in water accommodated fractions (WAF) less than 10 ppm, then no dispersant, no matter its individual toxicity, will meet the threshold values for listing.

Moreover, the creation of arbitrary thresholds for passing such tests, especially when EPA has shown that modern dispersants are of low toxicity, is not supportable and it is not clear whether any dispersant will be approved for the NCP Product Schedule when both toxicity and effectiveness tests are required. Specifically, setting an acute toxicity threshold of dispersants plus oil at an LC50 of 10 ppm has a high probability of eliminating an effective dispersant from the list, i.e., any dispersant that passes the baffled flask effectiveness test. We know from previous testing by industry that many (or even most) crude oils by themselves have LC50's below 10 ppm. Of more relevance is the use of toxicity results to assist in a NEBA-based discussion of the relative merit of using a particular dispersant in favor of another. Similarly, the attempt to link lab test results to decisions relating to pre-authorization and use is misplaced since laboratory tests are useful in understanding differences between products, not their real world performance.

The Proposed Rule indicates that the Agency " *does not state or imply* " that one response option is endorsed over another which is a practice shared by industry. However, the preponderance of additional requirements for dispersant u se will likely have the unintended consequence of prohibiting, or delaying the approval of, dispersant use in a timely manner regardless of whether they may be the best, or only, response option. Further, the language contained in the preamble suggests that the Agency considers dispersants as a last priority, rather than giving them equal consideration with all other appropriate response options. Additionally, API and NOIA are concerned that the Proposed Rule may cause chemical

[1] Scientific reasoning that the dispersant only contributes ~2.5% of the TPH toxicity is based on the following. In a 1:20 disp ersant-to-oil ratio (DOR), the dispersant proportion is 5%. The solvent component of modern day dispersants is roughly half, which mean s the overall TPH contribution from the dispersant is ~2.5%. As a result, the toxicity is dominated by the oil constituents.

manufacturers to either disco ntinue the production of effective, commercially -available products or discontinue research into the development of the next generation of dispersants.

A Net Environmental Benefit Analysis Process Ensures Response Choices that Minimize the Impacts of Oil Spills on People and the Environment

Any changes to the rule must take into account the Net Environmental Benefit Analysis (NEBA)[2] process. When a spill occurs, the OSC must make time-critical choices about which response options to implement immediately and effectively to manage the potential impacts of the spill. The NEBA process helps the OSC select choices that minimize the impacts of the spill on people and the environment. NEBA is a consensus-based planning tool that is used to bring natural resource trustees, regulators and stakeholders together to address resource-management decision-making needs for an oil spill response. It provides a structured, scientific method to evaluate the likely net reduction, or possible increase, in environmental and socio-economic impacts resulting from the implementation candidate response options as compared to the natural recovery or "do nothing" alternative. An assessment of the trade-offs associated with each option is then conducted taking into consideration the potential impacts to sensitive environmental resources and local community values. The NEBA process determines which response option or combination of options can best reduce the overall impact of spilled oil to common values and promote the fastest overall ecosystem recovery. Of particular note, the NEBA process considers the potential effects to a wide range of concerns, ranging from likely environmental effects (i.e., deep marine to terrestrial, etc.) to possible economic consequences. When initially considering the potential outcomes, no individual area receives a greater priority over another. Instead, all are weighed from an equal footing and only after evaluating all the likely effects is a determination made regarding the response option that offers that greatest environmental benefit. In so doing, decision-makers ensure that an unbiased process is used, with input from relevant stakeholders, to determine the best course of action. This process has been adopted all over the world to identify response methods that will yield the greatest benefit with the least impact.[3]

The NEBA process can be used in both oil spill preparedness planning and response. In the United States, a formal NEBA process is conducted during the planning phase at the Area Committee and Regional Response Team (RRT) levels with input from state and Federal participants to determine the benefits and limitations from using each response technology within their individual areas of responsibility. In most offshore regions of the U.S., this is accomplished in the form of the Consensus-Based Ecological Risk Assessments (CERA) process (NEBA equivalent) used by the U.S. Coast Guard (USCG)[4] and supported by the National Oceanic and Atmospheric Administration (NOAA), EPA, other Federal and state agencies, and academia. More than twenty workshops were held in various locations around the continental U.S., Caribbean, and Alaska from 1995 to 2011 to compare the benefits and risks of various response options when considering resource trade-off decisions. All of the workshops resulted in final publications (available from USCG) that were delivered to the Area Committees and RRTs to assist with response planning. One example of how this CERA/NEBA process was used to inform dispersant use decision-making is summarized in several papers authored by regulators in the state of California.[5] The applicability of the CERA/NEBA process as a tool for facilitating dispersant decision-making during spill response and planning was also evaluated by NOAA and published in a 2008 paper.[6]

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[2] NEBA is similar to the Consensus Ecological Risk Assessment (CERA) process developed by the U.S. Coast Guard in the mid -1990s.

[3] See, *Incident management system for the oil and gas industry: good practice guidelines for incident management and emergency response personnel*, IPIECA, August 2014, http://www.ipieca.org/publication/incident-management-system-oil-and-gas-industry-good-practice-guidelines-incident-manage.

[4] Aurand, D., Walko, L., & Pond, R. (2000). *Developing consensus ecological risk assessments: Environmental protection in oil spill response planning*. Washington, D.C.: U.S. Coast Guard.

[5] Addassi, Y. N., & Faurot-Daniels, E. (2005). California oil spill dispersant plan -Achievement through cooperation. *Proceedings of the 2005 International Oil Spill Conference, 2005* (1), 433 -437. doi:10.7901/2169 -3358-2005-1-433; Addassi, Y. N., Sowby, M.,

In 2014, the practice of incorporating a NEBA review during Source Control drills in the Gulf of Mexico was implemented by RRT VI during several large scale spill response exercises,[7] which has fostered good cooperation between the agencies to focus on the goal of minimizing shoreline community and ecosystem impacts by selecting response options that can maximize the effective treatment of oil offshore. NEBA discussions during both of these recent spill drills resulted in the concurrence of the RRT to authorize dispersants (both aerially and subsea) to treat the offshore oil slick.

Government decision-makers and/or the Responsible Party can conduct a rapid NEBA during actual spill responses to compare and rank the strengths and weaknesses of different response options for the incident-specific conditions. Lessons learned following a spill are incorporated into future policies and plans. Ecological recovery studies of the 1996 *Sea Empress* spill in the UK demonstrate the benefit of dispersant application along the coastline to prevent persistent oil from stranding in shoreline habits.[8] A more recent publication examined the recovery rates for a tropical ecosystem during a controlled study of dispersed versus untreated shoreline oil slicks and concluded that better recovery occurred in the areas where dispersant was used.[9]

Regrettably, the Proposed Rule lacks a NEBA perspective. It incorrectly focuses on intrinsic, hazard-based criteria for substance listing and ignores the more appropriate risk-based trade-off decision process responders must consider during an oil spill response. It does not recognize that dispersants are only applied when regulators and response decision-makers deem them necessary to prevent more significant and harmful consequences. The Proposed Rule favors protecting pelagic and benthic species while putting critical habitats, such as the ocean's surface and near-surface zones, as well as sensitive shorelines, at greater risk to environmental damage. Modeling results presented at the 2005 International Oil Spill Conference concluded that:

> *"…the results of the modeling support the contention that there are more opportunities to save wildlife, shorelines, and sensitive habitats along the shore with dispersant use than there are risks of impacting water column biota. If the areas that would be impacted by surface oil are those where wildlife are concentrated, as they typically are near shore, and the water column impacts resulting from dispersant use would be offshore where water column biota are lower in abundance, the trade-off is much more heavily weighted toward dispersant use before oil comes near the shoreline. In fact, for medium or smaller spills, or even larger spills where sensitive species and life stages are not present, the impact of dispersant use on water column biota could be negligible."*[10]

Parker-Hall, H., & Robberson, B. (2005). Establishment of dispersant use zones in the state of California: A consensus approach for marine waters 3 –200 nautical miles from shore. Proceedings of the 2005 International Oil Spill Conference, 2005(1), 187 -191. doi:10.7901/2169-3358-2005-1-187.

[6] Mearns, A., & Evans, M. (2008). Evaluating the consensus ecological risk assessment workshop as a tool for facilitating decision - making during spill response and planning. *Proceedings of the 2008 International Oil Spill Conference Proceedi ngs, 2008*(1), 739-741. doi:10.7901/2169-3358-2008-1-739

[7] Freeport McMoran Source Control drill (April 30-May 1, 2014) and BP Source Control drill (Nov 5-6, 2014)

[8] Lunel, T., Rusin, J., Bailey, N., Halliwell, C., & Davies, L. (1997). The net environmental benefit of a successful dispersant operation at the Sea Empress incident. *Proceedings of the 1997 International Oil Spill Conference, 1997* (1), 185 -194. doi:10.7901/2169-3358-1997-1-185

[9] Baca, B., Rosch, E., DeMicco, E. D., & Schuler, P. A. (2014). TROPIC S: 30 -year follow -up and analysis of mangroves, invertebrates, and hydrocarbons. *Proceedings of the 2014 International Oil Spill Conference, 2014* (1), 1734 -1748. doi:10.7901/2169-3358-2014.1.1734

[10] French McCay, et al., Modeling Fates and Impacts of Hypothetical Oil Spills in Delaware, Florida, Texas, California, and Alaska Waters, Varying Response Options Including Use of Dispersants. International Oil Spill Conference Proceedings: May 2005, Vol. 2005, No. 1 pp. 735-740. Also available at http://ioscproceedings.org/doi/pdf/10.7901/2169-3358-2005-1-735.

For oil spills where dispersant use is contemplated, a typical NEBA will predict the potential impacts to aquatic organisms from an increased exposure to dispersed oil droplets. Dispersant use is typically restricted to water depths greater than 10 to 20 meters so that dilution will minimize exposure or impacts to the benthic community. Studies have shown the dispersed oil concentrations are rapidly diluted within the large volume of receiving water from a high of a few hundred parts per million to a few parts per million within a few hours. Consequently, exposure to the toxic effects of the dispersed oil will be of short duration and limited to those aquatic organisms that inhabit near surface waters. The NEBA will also estimate the reduction of impacts to waterfowl, shoreline ecosystems and coastal communities resulting from the reduction in, or elimination of, the quantity of floating oil. The trade-offs of a short term increase in exposure of selected aquatic organisms to dispersed oil versus an increase in oil impacting coastal environments and communities must be evaluated and take into account local community values to determine if dispersant use is warranted.

Given the challenges of implementing other response options, a decision not to have dispersants available to responders could result in oil stranding on shorelines where it may persist for an extended time. Effectively dispersed oil will biodegrade rapidly so that the majority of the oil will persist for a much shorter period of time. Reducing the persistence of the oil in the environment is critical to reducing the overall impacts.[11] This is not to say that water column organisms are less intrinsically important than surface or shoreline species but using NEBA can and does provide a critical framework for deciding what is best for the overall ecosystem. Important considerations to understand in assessing this tradeoff include:[12]

- The impacts to the water column are mitigated by dilution – that is the potentially toxic levels of dispersed oil are limited in duration and physical coverage/extent and effected populations (e.g., plankton) tend to be ones that recover quickly.
- Oil effectively dispersed in the water column will generally biodegrade after a few days or weeks while oil stranded in near shore and shoreline areas can persist for years.
- Near shore and shoreline areas are the nursery grounds for many species, even those that live offshore as adults, so protecting the near shore areas also protects organism that live as adults offshore.
- The bio density of near shore areas (swamps and marshes) is far greater than the bio density in pelagic waters offshore.
- Dispersants also protect surface dwelling organisms offshore, i.e., marine mammals, birds, and turtles, through removal of floating oil.
- Slicks on the water surface are far more likely to impact fish eggs and larva that tend to float at or near the water surface than quickly removing oil from the surface with dispersants.
- In certain geographies, slicks can impact mangroves and other shoreline vegetation that can subsequently die off and result in shoreline erosion that increases sediments in the near shore water column and potentially affects corals and other species.

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[11] Hazen, et al., Deep-Sea Oil Plume Enriches Indigenous Oil-Degrading Bacteria, Science, 330 (2010), pp. 204-208.

[12] (a) French McCay, D., and Graham, E. Quantifying Tradeoffs – Net Environmental Benefits of Dispersant Use. International Oil Spill Conference Proceedings: May 2014, Vol. 2014, No. 1, pp. 762-775.
(b) McManus, M.A., Woodson, C.B. Plankton Distribution and Ocean Dispersal. The Journal of Experimental Biology, 215 (2012), pp. 1008-1016.
(c) DeMicco et al., Net Environmental Benefit Analysis (NEBA) of Dispersed Oil on Nearshore Tropical Ecosystems: TROPICS – The 25th Year Research Visit, International Oil Spill Conference Proceedings, May 2011, Vol. 2011-282.
(d) Varela et al., The Effect of the "Prestige" Oil Spill on the Plankton of the N -NW Spanish Coast. Marine Pollution Bulletin 53 (2006), pp. 272-296.
(e) Addassi, et al., Establishment of Dispersant Use Zones in the State of California: A Consensus Approach for Marine Waters 3-200 Nautical Miles from Shore. International Oil Spill Conference Proceedings: May 2005, Vol. 2015, No. 1, pp. 187-191.

The Agency should consider the NEBA process and how and why dispersants are used before making changes to the Subpart J Regulations.

The Proposed Rule has the Practical Effect of Treating Dispersants as Toxic and either Prohibiting or Discouraging the Use of this Critical Response Tool

Dispersants are an essential tool in the responder's toolkit. They are products used in oil spill response to dilute the oil in the water column to enhance natural microbial degradation, a process where microorganisms remove oil from the environment.[13] All marine environments contain naturally occurring microbes that feed on and break down crude oil. Dispersants aid the microbial degradation by forming tiny oil droplets, typically less than the size of a period on this page (<100 microns), thus increasing the oil's surface area and making it more available for the petroleum-degrading microorganisms to consume.[14]

All oil spill response options have their place in the response toolkit because of the extreme variability of spill conditions. Dispersants become a critical response tool for larger spills far from shore; spills more distant from stockpiles of recovery and containment equipment; when weather and ocean conditions preclude the use of other options; or when weather conditions are predicted to become more severe because they can be rapidly applied by aircraft or subsea (at a high encounter rate) when wind and wave conditions prevent other vessel-based recovery options.

The Proposed Rule focuses on the potential risks of dispersants without recognizing the benefits and core purpose of dispersants, which is to rapidly dilute the spilled oil and enhance removal of the oil from the environment through microbial degradation, resulting in reduced VOCs and shoreline impacts. Ultimately, the purpose of dispersants is to reduce the overall toxicity and persistence of oil in the environment.

We believe that research showing that dispersants do not enhance the biodegradation of oil from the environmental is based on studies that used unrealistically high concentrations of dispersed oil. Prince and Butler only recently developed a biodegradation test method that compares dispersed oil biodegradation to biodegradation of a surface slick at representatively low concentrations (i.e., 2.5 ppm).[15] This research unambiguously shows the enhanced biodegradation of dispersed oil over allowing an oil slick to remain on the surface. Most prior dispersed oil biodegradation studies used oil concentrations that were two or more orders of magnitude too high.[16]

Dispersants have proven to be an effective way to respond to an oil spill, most notably in those situations that prohibit, or otherwise limit, the effectiveness of other response measures. We submit that all response options should be contemplated based on their individual, or combined, ability to respond to

[13] American Society for Microbiology (ASM). (2011). *FAQ: Microbes & oil spills.* Retrieved from http://academy.asm.org/images/stories/documents/Microbes_and_Oil_Spills.pdf; Personna, Y., Boufadel, M., & Zhang, S. (2014). Biodegradation of dispersed Endicott oil in controlled experiments. Proceedings of the 2014 International Oil Spill Conference, 2014(1), 1126-1140. doi:10.7901/2169-3358-2014.1.1126; McFarlin, K., Leigh, M., & Perkins, R. (2014). Biodegradation of oil and dispersed oil by arctic marine microorganisms. Proceedings of the 2014 International Oil Spill Conference, 2014(1), 300-317. doi:10.7901/2169-3358-2014-1-300317.1.

[14] Hazen, T. C., Dubinsky, E. A., DeSantis, T. Z., Andersen, G. L., Piceno, Y. M., Singh, N., Jansson, J. K., Probst, A., Borglin, S. E., Fortney, J. L., Stringfellow, W. T., Bill, M., Conrad, M. E., Tom, L. M., Chavarria, K. L., Alusi, T. R., Lamendella, R., Joyner, D. C., Spier, C., Baelum, J., Auer, M., Zemla, M. L., Chakraborty, R., Sonnenthal, E. L., D'haeseleer, P., Holman, H. N., Osman, S., Lu, Z., Van Nostrand, J. D., Deng, Y., Zhou, J., & Mason, O. U. (2010). Deep-sea oil plume enriches indigenous oil-degrading bacteria. *Science*, *330*, 204-208. doi:10.1126/science.1195979

[15] Prince, R.C. and Butler, J.D. A protocol for assessing the effectiveness of oil spill dispersants in stimulating the biodegradation of oil. Environ Sci Pollut Res Int. 2014 Aug;21(16):9506-10. doi: 10.1007/s11356-013-2053-7. Epub 2013 Aug 13.

[16] Lee, K., Nedwed T., Prince R.C., Palandro D., Lab tests on the biodegradation of chemically dispersed oil should consider the rapid dilution that occurs at sea. Mar Pollut Bull. 2013 Aug 15;73(1):314-8. doi: 10.1016/j.marpolbul.2013.06.005. Epub 2013 Jun 27.

discharged oil, either on or below the water's surface, in a manner that causes the lowest negative effect on the environment and promotes the most rapid recovery of the affected ecosystem.

Laboratory Dispersant Efficacy Tests do not Predict How Well Dispersants Will Work in the Wide Range of Real World Spill Conditions

Although laboratory dispersant effectiveness testing can be used as a comparative screening tool for different dispersants under laboratory conditions, the Proposed Rule appears to suggest these results will be representative of actual effectiveness when applied to oil spills on open water. However, these results do not translate to the wide range of field conditions, oil types and weathering degrees in real world oil spills. Specifically, the static conditions in a lab, where researchers conduct experiments with known quantities of oil and dispersant, cannot be replicated at sea. Although the amount of dispersant that is to be applied is calculated based on a desired initial dispersant-to-oil ratio, the quantity and characteristics of oil during a real spill is an estimate, making any definitive, quantitative effectiveness estimation problematic. Localized determinations of the relative effect of surface dispersant application may be evaluated qualitatively using visual cues or semi-quantitatively using ultraviolet fluorometry, which is sufficient for operational effectiveness assessments. Additionally, much of the detailed spill monitoring specified by the Agency in the Proposed Rule has no bearing on dispersant effectiveness monitoring and cannot be conducted in a time-frame that will support day-to-day, near real-time operational decision-making.

We suggest the language in the Proposed Rule be modified to clarify the laboratory efficacy testing is for comparative purposes only and that the pass/fail criteria be established at a level that will only eliminate the least effective dispersants from being listed. Some dispersants that only exhibit average effectiveness in the laboratory tests may be very effective in certain situations in the field and the Response Community needs to maintain as many options as possible to select the most appropriate response option for a given set of conditions and circumstances.

Dispersant Toxicity Tests do not Accurately Simulate Real World Exposures.

API and NOIA acknowledge that there may be benefits to a so-called "gate-keeper" toxicity test with dispersant alone to ensure that any dispersants that might contain highly toxic components would not be accepted for listing on the Product Schedule. However, it is unclear what rationale was used for the dispersed oil toxicity tests stated in the Proposed Rule.

The relevant factors in dispersed oil toxicity include the concentration of dispersed oil, or oil compounds, in the water and the duration of exposure. The fundamental basis of aquatic toxicity testing is to create a set of concentrations and expose the animals for a duration where observable effects can be measured. As a result, laboratory exposures are useful for data collection and generating toxicity curves; however these conditions are not representative of the real world where the natural processes of rapid dilution and biodegradation act on the dispersed oil to dramatically reduce the exposure conditions [17]. Using acute 48- or 96-hour LC_{50} toxicity tests with dispersed oil mixtures (to calculate LC_{50} values) is considerably different from the likely exposure regime experienced by marine organisms in the open sea following an oil spill. For this reason, the Chemical Response to Oil Spills: Ecological Effects Research Forum (CROSERF)[18] was assembled to develop new dispersed oil toxicity testing protocols that account for rapid dilution. Academia, regulators and industry funded a ten year program through CROSERF to

[17] Lee, K., Nedwed, T., & Prince, R. (2011). Lab tests on the biodegradation rates of chemically dispersed oil must consider natural dilution. *Proceedings of the 2011 International Oil Spill Conference, 2011* (1), 245. doi:10.7901/2169-3358-2011-1-245
[18] Aurand, D., & Coelho, G. (2005). *Cooperative aquatic toxicity testing of dispersed oil and the chemical response to oil spills: Ecological effects research forum (CROSERF)*. Ecosystem Management & Associates, Inc., Technical Report 07-03. Lusby, MD.

develop new testing methods that are more aligned with real -world field c onditions. This data is now available via the NOAA Dtox database.[19]

Results from dispersed oil monitoring during the Deepwater Horizon (DWH) spill is clear evidence of the rapid dilution occurring after surface dispersant application [20] and subsea dispersant injection .[21] Further evidence of the rapid dilution of dispersed oil in the field is found in the analytical results from several field trials conducted in the North Sea to characterize the fate and transport of dispersants applied to an open ocean oil slick. These field experiments and the subsequent chemical characterization condu cted in 1994,[22] 1995,[23] and 1996[24] showed a rapid (within one hour) dilution of dispersed oil concentrations following dispersant application. This same effect was observed during the DWH response following surface dispersant application.[25] Since the proposed EPA dispersed oil testing does not take into account dilution, it is more likely that reported toxicity values at a rtificially high laboratory concentrations, not representative of real -world dilution exposures, will lead to misinterpretation of these tes t results and possibly preclude the use of dispersants altogether.

We strongly suggest that the laboratory toxicity test be limited to dispersants only to screen out the more toxic products and that toxicity testing for dispersed oil be explicitly elimin ated from the Rule. It is very likely that all dispersants will be unable to pass the dispersed oil toxicity test as existing oil toxicity data indicates the test oils alone will be unable to pass the test.

In the context of NEBA (and its forebear, th e Consensus Ecological Risk Assessment, or CERA), scientists understand and accept that marine organisms may experience localized, transient toxic effects from exposure to dispersed oil, but these local effects are offset by the benefit of dispersant use i n preventing or minimizing oil drifting ashore and causing severe and long -term harm to more sensitive coastal resources and communities . While knowledge of the potential for toxic effects to marine organisms is useful for conducting a NEBA, it cannot be t he sole indicator that dictates the use of dispersants. All resources -at-risk as well as coastal communities must be considered in the response option decision-making process, with the emphasis on coordinating a response that will result in the least overall impacts while promoting the most rapid overall recovery of the affected ecosystem.

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[19] Bejarano, A., Chu, V., Dahlin, J., & Farr, J. (2 014). Development and application of dtox: A quantitative database of the toxicity of dispersants and chemically dispersed oil. *Proceedings of the 2014 International Oil Spill Conference* , *2014*(1), 733 -746. doi:10.7901/2169-3358-2014.1.733

[20] Operational Science Advisory Team (OSAT 1). (2010). *Summary report for sub-sea and sub-surface oil and dispersant detection: Sampling and monitoring. Unified Area Command, New Orleans*. Retrieved from: http://www.dep.state.fl.us/deepwaterhorizon/files2/osat_report_17dec.pdf .

[21] Coelho, G., Aurand, D., Essex, L., Parkin, A., & Robinson, L. (2012). *Monitoring subsurface dispersant injection during the MC252 incident, volume 1*. Lusby, MD, USA: Ecosystem Management & Associates, Inc.

[22] AEA Technology. (1994). *International calibration of laboratory dispersant test methods against sea trials* . Oxfordshire, UK: AEA Technology.

[23] AEA Technology. (1995). *International calibration of laboratory di spersant test methods against sea trials. Field trial report. July 1995 sea trials*. Oxfordshire, UK: AEA Technology ; Jones, M., & Petch, S. (1995). A report on the analysis of hydrocarbons in sea waters and associated samples from trial oil spills off east ern England, July 1995. Newcastle upon Tyne, UK: University of Newcastle upon Tyne.

[24] Strøm-Kristiansen, T., Hokstad, J. N., Lewis, A., & Brandvik, P. J. (1997). *NOFO 1996 oil on water exercise – analysis of sample material. SINTEF Data report number STF66 A97050*. Trondheim, Norway: SINTEF ; Coelho, G. M., Aurand, D. V., Petch, G.S., & Jones, D.M. (1998). Toxicity bioassays on dispersed oil in the North Sea: June 1996 field trials. Ecosystem Management & Associates, Inc., Report 96-02. Purcellville, VA: Ecosystem Management & Associates, Inc.

[25] Huber, C. A., Steen, A., & Parscal, B. (2014). Does wave height matter for effective surface dispersant application? *Proceedings of the 2014 International Oil Spill Conference, 2014* (1), 747-761. doi: 10.7901/2169-3358-2014.1.747

During an Oil Spill, Operational Monitoring Should Focus on How the Selected Response Options Mitigate the Effects of the Oil on the Environment

Following an oil spill, the primary emphasis of operational response monitoring should be directed toward the response effort and the effect that those efforts have on reducing the net negative environmental consequences. The Agency proposes the requirement to "*monitor agent use in the environment*." API and NOIA submit that operational monitoring should not be focused on monitoring the agent (i.e., dispersant) use, but rather on monitoring the overall changes to the characteristics, transport, and fate of the oil within the ecosystem. We also submit the Agency should acknowledge that the primary issue in a spill scenario is oil in the environment (not dispersants in the environment) and that operational monitoring efforts should concentrate on evaluating how well the response option (or options) mitigates the negative effects of the oil on sensitive environmental resources.

EPA proposes monitoring that will provide a comprehensive assessment of the environment. However, the monitoring actions required in the Proposed Rule go beyond those required to inform operational decision-making and cross into the realm of Natural Resource Damage Assessment (NRDA) monitoring. One reason for this is that NRDA monitoring requires days to years before opinions on damages are made. Operational decisions require real or near-real time data. Damage assessment monitoring should not fall under the purview of spill response decision-making, and any monitoring of response options should be done solely to ensure the effectiveness of that response technique.

The Proposed Rule includes additional surface dispersant effectiveness monitoring requirements beyond those established in current practices, with no justification of how this additional monitoring burden will enhance operational response decision-making. Further, the additional requirements do not differentiate between surface and subsurface use of dispersants. As a result, the Proposed Rule would apply an unnecessary set of requirements in the event of surface dispersant usage in lieu of those monitoring techniques outlined in the USCG *et al., Special Monitoring of Applied Response Technologies* (SMART) protocols.[26] Additionally, the monitoring regime outlined in the Proposed Rule deviates substantially from that recommended by the National Response Team (NRT) in its *Environmental Monitoring for Atypical Dispersant Operations: Including Guidance for Subsea Application and Prolonged Surface Application,*[27] which also advocates the adaptation of the SMART monitoring regimen. These additional monitoring requirements, if adopted, may divert valuable resources away from the response effort, potentially prolonging the spill event. They may also preclude the use of dispersants in many cases if required to be in place as a condition of authorization, as it can take many days to assemble the equipment, vessels, and scientific experts.

During a subsea dispersant application, the Agency proposes an exhaustive set of environmental monitoring parameters that include *in situ* oil droplet size distribution analysis, *in situ* fluorometry, fluorescence signatures, dissolved oxygen, total petroleum hydrocarbons, carbon dioxide, methane, heavy metals, turbidity, water temperature, pH, and conductivity. As stated previously, we concurs with monitoring that supports effective, operational decision-making, but argues that most of these parameters will not provide any information on the effectiveness of the subsea dispersant application, will unnecessarily require additional scientific resources and will delay the reporting of key operational monitoring parameters. Spill response effectiveness monitoring should be based on simple, rapid assessments that can provide near real-time information on the progress of the response. For this reason, we believe that any mandated operational monitoring should be both incident-specific and designed to integrate into the other operational activities taking place. As such, the Responsible Party

[26] These protocols were jointly developed by the U.S. Coast Guard, National Oceanic and Atmospheric Administration, U.S. Environmental Protection Agency, Centers for Disease Control and Prevention, and Minerals Management Service.
[27] See http://www.nrt.org/production/nrt/nrtweb.nsf/AllAttachmentsByTitle/SA-1086NRT_Atypical_Dispersant_Guidance_Final_5-30-2013.pdf/$File/NRT_Atypical_Dispersant_Guidance_Final_5-30-2013.pdf?OpenElement

shares in the responsibility with the OSC in determining what monitoring needs to be done in ac cordance with the region -specific pre -approval and planning process. Comprehensive, detailed environmental monitoring is not an appropriate topic for this Proposed Rule.

The Agency's Economic Review Fails to Recognize Significant Costs of the Proposed Rule.

The Regulatory Impact Analysis (RIA) conducted by the Agency does not estimate all of the potential policy scenarios, and the economic impacts of those policies, that are issued in the Proposed Rule. Therefore, the RIA neglects to provide the public with a full description of the proposal's impact. Additionally, EPA did not provide any quantified social benefits that could result from implementation of any part of the Proposed Rule. An independent assessment [28] of the proposal (see Attachment A) has found that including the impacts of the requirements neglected by the Agency in the RIA would increase the cost of the proposal three times. In addition to underestimating compliance costs and administration costs for both industry and the federal governmen t, the RIA supporting the proposal fails to calculate the costs associated with the immediate market imbalances that could occur if products are removed from the market, either temporarily or permanently.

Beyond the market imbalance, removal of dispersants from the toolbox could result in additional clean-up costs that are not considered in the RIA. As noted in Attachment A, "To the degree this rulemaking reduces chemical and biological agent availability and innovation, the nation will have less tools av ailable to fight oil spill damage." A possible outcome of this limitation "is that future oil spill damages would be more severe than they would be if the rule did not occur." EPA did not quantify the costs associated with the possibility that the rule w ould limit the ability of the Response Community to keep oil from reaching shorelines or damaging ecosystems to a greater extent than if dispersants were readily available. We believe additional costs could extend to costs associated with environmental cla ims, increased response costs due to the need for shoreline cleanup, and other costs of environmental damages to sensitive shorelines, birds, and lost economic and recreational opportunities. In fact, EPA confuses the social benefits of avoiding an oil spill or having an appropriate response to a spill with the social costs of a spill, such as lost tourism revenue and other environmental damages. Furthermore, the Agency underestimated the costs to small businesses, which resulted in the false conclusion t hat small businesses would not be materially affected by the proposed ruling. Using realistic estimates, the attached third-party report finds that 43 percent of the firms identified by the Agency would face compliance costs in excess of three percent of a nnual sales and risk of closure. The Agency should amend the methodology and deficiencies found in the RIA, as discussed here and in the attached report.

We explicitly incorporate the comments and recommendations addressing the RIA provided by the independent assessment (see Attachment A).

The Proposed Rule Conflicts with Current Facility and Vessel Oil Spill Response Plans

Effective September 30, 2009, facility and tank vessel plan holders were required to update their response plans to include planning for responding to discharges using dispersants under a USCG rulemaking. Specifically, dispersant equipment is required in areas where it has been pre-determined that dispersants would be a viable oil spill mitigation technique and pre-authorizations have been established. Previously, with regard to dispersants for facilities located in areas with pre-authorization, a mechanical recovery equipment credit for up to 25% could be obtained for certain persistent oils.

[28] The independent assessment was prepared by the Policy Navigation Group (PNG), which consists of a diverse team of senior policy analysts with experience at Office of Management and Budget in the White House, managers, economists, engineers, lawyers and skilled researchers, with particular expertise in both private and public sector environmental management strategies including benefit-cost analysis and life-cycle assessments. See http://www.policynavigation.com/index.php?page=bios for more information about PNG.

However, since 2009 facilities handling certain persistent oils in areas with pre-authorization must ensure, through contract or other approved means, the availability of dispersant response resources and the commencement of dispersant-application operations within 7 hours of the decision by the OSC to use dispersants. Dispersant response resources must include: (1) sufficient volumes of dispersants to meet specified regulatory requirements, (2) dispersant application platforms and systems sufficient to meet planning standards, and (3) trained personnel.

Similarly, vessels transporting certain persistent oils in areas where dispersants are pre-authorized have been since 2009 required, through contract or other approved means, to have dispersant-related response resources meeting dispersant planning standards, including appropriately trained dispersant-application personnel. The dispersant resources must include the identification of each primary dispersant staging site used by each dispersant-application platform and the corresponding distance to the associated stockpile. Similar requirements were adopted for non-tank vessels in 2013.

There are also aerial tracking requirements under the current response plan requirements, and there are now pre-authorization zones for the use of dispersants outlined in the applicable Area Contingency Plan throughout the nation for every state except for Alaska and Connecticut whereby the OSC has the authority to expedite the authorization for the use of dispersants. Although there are no specific dispersants requirements under Bureau of Safety and Environmental Enforcement's (BSEE) response plan requirements for offshore facilities, it is clear based on experience with the *Deepwater Horizon* that dispersants are a key response tool that must be available on an expedited basis to respond both to a spill on the surface of the water and subsea in case of a blowout.

The Proposed Rule, however, appears to directly conflict with Coast Guard response plan requirements and raises a host of questions related to the continued validity of facility and vessel response plans. It is unlikely that any dispersants will pass both the efficacy and toxicity tests required by the Proposed Rule as a precursor to being listed on the NCP Product Schedule. For example, the EPA proposal may have the practical effect of virtually eliminating the entire stockpile of available dispersants nationwide that will be able to be listed on the NCP Product Schedule following full implementation of the Proposed Rule. In addition, it could have the practical effect of eliminating the usefulness of preauthorization zones. This could result in invalidating response plans.

Furthermore, the Proposed Rule, if implemented , would require significant additional requirements for information needed an d monitoring required before dispersants use approval is granted , which may prevent a plan holder from being able to deploy dispersants in the desired/required amount of time. For some spills it may mean that dispersants cannot be used at all due to limit ed window of opportunity. Thus, the Proposed Rule raises significant issues related to USCG and BSEE facility and vessel oil spill response plan compliance and could result in unnecessary and increased environmental damage and corresponding spill claims a s a result of increased damages. In this regard, we note that although the President delegated to EPA the responsibility for the amendment of the NCP, the Proposed Rule may extend beyond the limits of EPA's authority under that delegation or the applicabl e law. Furthermore, amendments to the NCP must be coordinated with members of the NRT , including the USCG and other relevant agencies, prior to publication for notice and comment. 40 C.F.R. § 300.2. It appears this coordination may not have been accompl ished sufficiently to avoid the conflicts discussed above. Accordingly, we recommend that EPA further coordinate with the USCG and BSEE to ensure that the Proposed Rule does not adversely affect USCG and BSEE response planning procedures and requirements.

<u>Agency Review of Public Comments</u>

Due to the significance of this rulemaking to the industry and to other Federal government agency stakeholders involved with spill response, it is critical the Agency provide adequate time for these other

agencies to review and provide feedback on comments submitted by the public for EPA's benefit and consideration. Accordingly, we strongly urge the Agency to seek and consider comments from the other relevant Federal government agencies on the public's comments befor e it decides on how to handle all input received in response to the Proposed Rule.

Availability of EPA Testing Data

It has come to the attention of API and NOIA that the Agency may have testing data related to dispersants that may not have been released to the public. If that is the case then the Agency should make that data publicly available as soon as possible in some form of a supplemental notice to this Proposed Rule. Supplemental comments should then be requested to ensure that the public has an opportunity to provide additional informed comments for EPA's consideration before finalizing the Proposed Rule.

DETAILED COMMENTS ON THE PROPOSED AMENDMENTS TO 40 CFR PARTS 110 AND 300

Our specific section -by-section comments on the Proposed Rule are presented as follows . These comments conform to the organization of the Proposed Rule on a section-by-section basis.

I. § Section 300.5 Definitions

A. *The definition of burning agents should not include ignition devices*.

Burning Agents. The proposed defi nition reads: " *Burning agents are additives that improve the combustibility of the materials to which they are applied through physical or chemical means* ." In the preamble (80 Fed. Reg. 3385) an example is given of a helitorch system, stating that the mate rials in the helitorch would be considered burning agents. Additionally, the Agency is seeking comment on whether it should add ignition devices to the definition of burning agent.

The proposed defin ition seems to combine burning agents (materials that actually change the combustibility of the material they are added to) and ignition agents (materials used to start combustion, as is the case with the helitorch). This confuses the question around including ignition devices within the definition of burning agents which, if happens, could lead to administrative pauses in requesting a test burn, or any burn during an emergency phase of a response.

Igniters are used to initially heat a small portion of the oil to release vapors that can be ignited. The initi al process of ignition is not the same as improving overall combustibility of an oil. For example, when starting a campfire, a match does not change the combustibility of the wood; it simply provides the energy to start the combustion process. Similarly, a n in-situ burn ignition system provides energy to heat and vaporize some of the oil so that it can start to combust. For this reason, we submit that ignition devices (both ignition liquids and equipment) should not be included in the definition of a burning agent.

We recommend the Agency adopt the following definitions:

> *Burning agents are additives that change the combustibility of the materials to which they are applied through physical or chemical means prior to ignition.*
>
> *Ignition agents are additives that ignite, or initiate combustion of the materials to which they are applied.*

B. *EPA's definition of dispersants does not fully define dispersants.*

Dispersants. The EPA proposed definition of dispersants in 300.5 should be expanded to include an explanation of why the production of smaller droplets is important. The following language should be added to the end of the proposed definition: *"The formation of small droplets is intended to allow the oil to be rapidly diluted in open ocean environments t o non -harmful concentrations and to promote rapid biodegradation of the oil by naturally occurring microbes."*

C. *Oil-mineral aggregate ("OMAs") are not sorbents*

Sorbents. API and NOIA offer several comments on the definition of sorbents in 300.5, as follows:

- We support the addition of clay as an example of an inorganic/mineral compound, especial ly since commercial products made from clay are currently on the market.
- The Agency expresses concern that sorbents made from particulate matter may be used as sinking agents. Many inorganic/mineral compounds that are quite dense are commonly used on terrestrial spills on roads, pavements and other hard surfaces. The rule should not limit their use for these situations.
- Very fine clays (like those used in OMAs), wh ile still nominally denser than water, will still suspend in the water column because the particle sizes are very small. The rule needs to acknowledge that OMA's are not sinking agents. If the Agency is concerned that inorganic/mineral sorbents are being u sed as sinking agents, then their use should be explicitly prohibited for use as a sinking agent. Uses for other purposes should be allowed.
- EPA should not limit how sorbents in particulate form are used. The use of OMA technologies is not to sorb oil but to promote dispersion and biodegradation. The purpose of the technique is not the same as for sorbents and should not be regulated as such. OMA technologies should be excluded in the definition of sorbents and should have separate testing/authorization requirements.
- While most sorbents are collected and recovered from the environment, some natural organic sorbents are spread over oil on habitats where the oil cannot be easily removed. The sorbent ties up the oil to reduce further migration and to limit the exposure to flora and fauna. The Agency should not discourage this application as it is sometimes considered a "best practice" for certain oiled habitats. As such, the Agency should either remove the phrase, "generally collected and recovered…" or add lang uage to indicate that loose, organic sorbents used in wetlands may not be recovered.

D. *Saltwater should be defined in the Proposed Rule*

Saltwater. In column 2, 80 Fed. Reg. 3389, of the preamble and subsequent sections the term "saltwater" is used. It is recommended that saltwater be defined in the definitions in 300.5 since the Proposed Rule states that dispersants are only allowed in saltwater environments. There is no definition of saltwater offered in the proposal. Dispersants have been shown to work a t salinity much lower than the typical 35 ppt ocean concentrations and effectiveness at salinity as low as 14 ppt has been reported. A definition where dispersants can be used needs to be provided, taking into consideration the data on dispersant performance in brackish or estuarine environments.

II. <u>Section § 300.910 Authorization for Agent Use</u>

A. *The Unified Command (UC) determines the best use of response assets on a daily basis.*

EPA should not establish a policy or rule that mechanical recovery is always the priority choice of countermeasures to use in a response. In the preamble for this section (column 1, 80 Fed. Reg. 3387), the Agency makes the following statement:

> *"During the DWH response, a priority countermeasures scheme was established to first use mechanical recovery via skimming/booming or in-situ burning followed by subsea dispersant and lastly surface dispersant use."*

This may have been the conceptual prioritization from the perspective of EPA, but in reality dispersants played a much larger role and it was the responsibility of the UC to determine the best use of response assets on a daily basis to account for weather conditions, VOC suppression, oil slick locations, response equipment availability, etc. Every spill is different and specific references to a single spill event can indirectly and inappropriately be perceived as a recommended practice and should be avoided. In any spill scenario, all response tools must be used to their maximum effect. The Agency's unprecedented use of "Directives" during the DWH incident severely limited the use of dispersants and circumvented the NEBA decision-making process allowing significantly more oil reaching the water's surface and, subsequently, the shoreline. This resulted in additional environmental impacts with no documented benefits to the marine ecosystem.

The Agency should reconsider this statement and prioritization as it puts unneeded complexity into the response decision-making process and inappropriately prioritizes mechanical recovery over other response options. The effectiveness of mechanical recovery can be limited in large, offshore spills as a result of typical environmental conditions, such as wave action/height or wind conditions as well as the extensive surface area that must be covered. Further, the Agency's artificial prioritization of this technique over other viable response options is contrary to the NEBA concept to determine the best blend of response options (with no pre-determined or pre-existing bias) to achieve the best overall protection and recovery for the ecosystem.

B. *Decision-making plans are also important to response decision-making.*

Later in the same paragraph referenced above, the Agency makes the following statement:

> *"The Agency believes that preauthorization or expedited decision making plans are critical elements of contingency planning activities."*

We strongly agree that preauthorization and expedited decision-making are important.

C. *Proposed changes involving preauthorization planning will increase the workload of all Regional Response Teams (RRTs) and Area Committees (AC) and restrict their flexibility.*

EPA should not restrict RRT or OSC activities related to preauthorization plans in areas where preauthorization is desirable and applicable. At the end of the same paragraph (column 2, 80 Fed. Reg. 3387), the Agency makes the following statement:

> *"The Agency believes these proposed revisions to the authorization of use provisions will assist OSCs, RRTs, and ACs in their advanced planning activities as they consider response methods that result in the greatest environmental protection."*

We agree with the intent to assist the OSCs and RRTs. However, the added complexity resulting from the Proposed Rule in authorization, monitoring and product listing, in addition to the overly conservative assumptions about dispersant effects described in the Proposed Rule, will not assist these entities.

Instead, it complicates and limits their ability to respond. The Agency must provide specific examples to show how this rule assists responders, particularly in relation to the OSC duties of making timely operational decisions, especially those decisions based on using the concept of NEBA to guide a response.

The existing rule states that the RRTs and Area Committees should evaluate the desirability of using various agents as part of their planning processes and state that "*RCPs and ACPs shall, as appropriate, include applicable preauthorization plans....*" Under the current rule, most RRTs did, in fact, create preauthorization plans, and those plans did include appropriate oceanographic and geographic limits. In practice, only Region VI used those plans to mitigate oil spills, and the Region VI OSCs, RRTs, and Area Committees became very familiar with their use. Policies and practices for the use of the RRT VI preauthorization plan were created by the appropriate member agencies without regulatory requirements and evolved to account for changes in information technology.

The proposed language states, "*RRTs and Area Committees shall address in a preauthorization plan…*" This becomes a mandate for all RRTs and Area Committees, including those areas that may have little interest or need for such plans. In fact, some RRTs and Area Committees do not even have water bodies where dispersants could be used. Most of the factors that are required to be considered by the Proposed Rule are already addressed in preauthorization plans (e.g., water depth, distance from shorelines). Some factors, however, such as the requirement that "*preauthorization plans must specify limits for the quantities and the duration of use*," for agents are new and have the effect of limiting the application and effectiveness of the pre authorization plans. This requirement would suggest that RRTs must know in advance the size of the potential releases, the resources at risk and what strategies are available to protect them, the dispersability of the oil, and the optimum dispersant-to-oil application rate. The OSC's flexibility and the effectiveness of the response method are limited and not assisted by such requirements, and EPA has not justified that the proposed requirement is necessary in relation to existing requirements.

The existing rule also states that "*preauthorization plans may address factors such as ...*" However, the Proposed Rule states "*Preauthorization plans should document…*" and adds requirements for consideration of factors such as "inventory" and "manufacturing capability of available agents." Again, this requires additional effort on the part of RRTs and Area Committees, where such work is already typically being done by industry as part of their planning processes.

The net effect of most of the changes involving preauthorization planning is an increase in the workload of all RRTs and Area Committees while restricting their flexibility in areas where preauthorization is desirable and applicable.

III. Section § 300.910(a) Use of Agents Identified on the Schedule on Oil Discharges Addressed by a Preauthorization Plan

A. *Adding more entities complicates the planning process.*

Section § 300.910(a) states that the RRTs and Area Committees shall address preauthorization plans. API and NOIA do not see any valid reason to add more entities to the planning process. In the preamble to this section, the Agency makes the following statement (column 1, 80 Fed. Reg. 3388):

> *"Therefore, the Agency is proposing to include RRTs as another planning entity with responsibility for developing preauthorization plans, and is requesting comment on this change, and on the advantages or disadvantages of keeping the development of these plans at the AC level."*

We believes that while the RRTs may provide more practical experience to the planning effort, the potential benefit would be overshadowed by the added complexity and bureaucracy. Unle ss the Agency can provide a sound rationale, we recommend against including the RRTs in the development of preauthorization plans.

B. *Area Committees should identify shoreline habitats that can benefit from the appropriate use of dispersants.*

In this same section of the preamble (column 3, 80 Fed. Reg. 3388), the Agency makes the following statement:

> *"The RRT's and ACs should identify the affected biological resources and their habitats likely to be negatively impacted, as well as those that are expected to benefit."*

API and NOIA support the requirement that A rea Committees should identify shoreline habitats that can benefit from the appropriate use of dispersants or other response methods. This will advance the use of the NEBA concept. Note that many of these resources are already identified in Area Contingency Plans.

C. *Preauthorization plans should provide the OSC sufficient authority to use chemical or biological response agents.*

In this same section of the preamble (column 1, 80 Fed. Reg. 3389), the Age ncy makes the following statement:

> *"The Agency is clarifying that if the preauthorization plan is approved in advance for chemical or biological agent use under specified discharge circumstances, then the OSC may authorize the use of the agents on the Sc hedule for their intended purpose without the incident specific concurrences and consultations described in paragraphs (b) of this section unless otherwise directed by the Administrator in accordance with current concurrence authority."*

We believe that the preauthorization plan should provide the OSC sufficient authority to use chemical or biological response agents. Since the Agency will have already been involved in the preauthorization plan approval, additional Administrator concurrence is unnecessary and would unduly complicate the response.

D. *RRTs and A rea Committees should be required to address and document their review and conclusions on whether or not the use of chemical or biological agents is appropriate.*

The Proposed Rule reads that:

> *"RRTs and Area Committees shall address in a preauthorization plan, as part of their planning activities, whether the use of chemical and biological agents listed on the Schedule on certain oil discharges is appropriate."*

This appears to compel RRTs and A rea Committees to document when and why such agents are appropriate, but does not require documentation or the rationale if they are not deemed appropriate, leaving the option open for RRTs and A rea Committees to simply not address prea uthorization for use of such agents. RRTs and A rea Committees should be required to address and document their review and conclusions on whether or not the use of chemical or biological agents is appropriate. API and NOIA suggest modifying the proposed language to read:

RRTs and A rea Committees shall address in a preauthorization plan, as part of their planning activities, whether or not the use of chemical and biological agents listed on the Schedule on certain oil discharges is appropriate and document the rationale for whether or not the use of an agent is considered appropriate.

IV. Section § 300.910(a)(1) Preauthorization Plan Development

A. *Dispersant use should not be limited by the preauthorization plan.*

EPA should not unnecessarily limit the use of dispersants through new req uirements in developing preauthorization plans. In this section (column 3 , 80 Fed. Reg. 3422), the Agency makes the following statement:

> *"For discharge situations identified where such agents may be used, the preauthorization plan must specify limits for the quantities and the duration of use* (Ed. emphasis added), *and use parameters for water depth, distance to shoreline, and proximity to populated areas."*

We assume that this provision in the Proposed Rule is a direct response to recommendation C5 on pag e 270 of the January 2011 report *Deep Water: the Gulf Oil Disaster and the Future of Offshore Drilling* by the National Commission on the BP *Deepwater Horizon* Oil Spill and Offshore Drilling. However, in the four years since the publication of this report, there have been numerous scientific, peer -reviewed environmental and health publications that have examined the dispersant use during DWH.[29] Subsequent laboratory toxicological studies of dispersed oil in the water column after dispersant application showed that there was no detectable harm to the species tested. Key findings from this research program included the following:

- Following surface dispersant applications, the dispersed oil c oncentrations in the water column return within 3 hours to background l evels due to vertical and horizontal dilution into surrounding ocean waters;

[29] Atlas, R. M., & Hazen, T. C. (2011). Oil biodegradation and bioremediation: A tale of the two wo rst spill in U.S. history. *Environmental Science & Technology, 45* (16), 6709-6715. doi:10.1021/es2013227; Chakraborty, R., Borglin, S. E., Dubinsky, E. A., Anderson, G. L., & Hazen, T. C. (2012). Microbial response to the MC -252 oil and Corexit 9500 in the Gulf of Mexico. Frontiers in Microbiology, Ecotoxicology and Bioremediation, 3, 357 -367. doi:10.3389/fmicb.2012.00357; Lubchenco, J. (2010, December 29). Oil spill clarifies road map for sea turtle recovery. The Miami Herald. Retrieved from http://www.nmfs.noaa.gov/mediacenter/docs/2011/mar/oil_spill_clarifies_road_map_for_sea_turtle_recovery.pdf ; Louisiana Department of Wildlife and F isheries. (2010, July). Oyster stock assessment report of the public oyster areas in Louisiana. Oyster Data Report Series No. 16. Retrieved from https://www.wlf.louisiana.gov/sites/default/files/pdf/page_fishing/32695 -Oyster%20Program/2010-oyster-stock-assessment-report.pdf; Operational Science Advisory Team. (2010). Summary report for sub-sea and sub -surface oil and dispersant detection: Sampling and monitoring. Retrieved from http://www.restorethegulf.gov/sites/default/files/documents/pdf/OSAT_Report_FINAL_1 7DEC.pdf; Operational Science Advisory Team. (2011). Summary report for sub -sea and sub -surface oil and dispersant detection: ecotoxicity addendum. Retrieved from http://www.restorethegulf.gov/sites/default/files/u306/FINAL%20OSAT%20Ecotox%20Addendum.pdf ; Shea, D. (2014, August 15). In re oil spill by the oil rig "Deepwater Horizon" in the Gulf of Mexico, on April 20, 2010. Expert report of Dr. Damia n Shea. New Orleans, LA: United States District Court, Eastern District of Louisiana. Retrieved from http://mdl2179trialdocs.com/releases/release20150120070000/TREX -013444.pdf; Soniat, T. M., King, S. M., Tarr, M. A., & Thorne, M. A. (2011). Chemical and physiological measures of oysters (Crassostrea virginica) from oil -exposed sites in Louisiana. Journal of Shellfish Research, 30(3), 713 -717. doi:10.2983/035.030.03 11; and U.S. Food and Drug Administration. (2010, October 29). NOAA and FDA announce chemical test for dispersant in Gulf seafood. Retrieved from http://www.fda.gov/NewsEvents/Newsroom/PressAnnouncements/2010/ucm231653.htm.

- The surface dispersants were applied over an operating area of 18,000 sq. mi., as a result, surface dispersant application rarely occurred in the exact same location, which further enhanced dilution of dispersed oil in the Gulf; and
- The mixing of relatively small volumes of dispersed oil into the very large, well -mixed Gulf of Mexico resulted in very short exposure durations of animals

These studies, conducted during and after the DWH aerial dispersant operations, indicate that there is no need to specify limits for the quantities and the duration of use of dispersants in a preauthorization plan. We believe the Section 300.910(a)(1) proposed requirement that specifies limits for the quantities and duration of use should be deleted from the Proposed Rule.

If the Agency wants to determine whether and how to limit dispersant use, we propose a joint EPA/industry study of existing research results to provide scientific credibility to any limitations that are established. Without scientific justification, artificially -set limitations on dispersant use would threaten the intent of using NEBA to support response decision -making and would limit the OSC's ability to make the best overall decision concerning the actual scenario and the environment.

To be clear, API and NOIA believe the amount of dispersant used should be based on the outflow of oil and the dispersant-to-oil ratio (DOR) that is effective in dispersing the plume. It is under stood that after a specified time period (e.g., 48 hours), the OSC will request RRT concurrence for continued dispersant use (except when there is a threat to human life and health) wherein concerns over quantities and duration can be evaluated.

We believe that the proposed preauthorization requirement to provide distance to shoreline and proximity to populated areas is not justified as the RRTs/Area Committees have already set these requirements in existing preauthorization plans as appropriate . Each RRT has developed preauthorization maps which clearly depict minimum distances from shore and minimum water depths. In addition to these already established distance and depth limitations, during an actual response, dispersant responders work with the RRT, OSC, and Responsible Party to set standard requirements and operating procedures for aerial spray safety setbacks of 1,000 ft. or more from vessels, platforms, response personnel, and sensitive marine life. Because these practices are already in place, there is no need to specifically regulate new parameters. For these reasons, API and NOIA believe the parameters for distance to shoreline and proximity to populated areas should be deleted.

We suggest that the Agency confer with other Federal agencies involv ed in oil spill planning and response and revise the language in the Proposed Rule to specify that preauthorization should rely on the concepts of CERA/NEBA as the foundation for pl ans and response decisions, not on artificially set limits on dispersant volumes. The USCG has conducted more than a dozen CERAs to help inform the decision-making process, but the Proposed Rule appears to contradict the CERA/NEBA tenets that were previously used to define preauthorization zones.

B. *EPA should work with other agenc ies to implement changes to preauthorization plans.*

Finally, since the Proposed Rule suggests that the Agency is adjusting the natural resource trustee concurrence requiremen t for preauthorization plans, we believe strongly that the Agency must demonstrate how it will work with other agencies to implement these proposed changes into the existing RRT preauthorization process and should specify a time interval for the relevant agencies to complete this integration (e.g., 1-year).

C. *The USCG and Bureau of Sa fety and Environmental Enforcement (BSEE) already regulate logistical aspects of dispersant use.*

There is no reason for the Agency to regulate logistical matters related to dispersant usage because they are already regulated by other agencies. In Section § 300.910(a)(1), the Agency makes the following statement:

> *"...and logistical factors including inventory, storage locations and manufacturing capability of available agents, availability of equipment needed for agent use, availability of adequately trained operators, and means to monitor agent use in the environment."*

There have been many advances in regulations and spill response planning in the past decad e. The regulations should acknowledge that there are now requirements in 33 CFR Parts 154 and 155 that have established a surface dispersant capability throughout the U.S. to apply up to 55,000 gallons (daily) of dispersant out to 50 nm offshore. These new regulations resulted in the development of 44 strategically located dispersant stockpile bases. Fur ther, these regulations require training and exercising of equipment and personnel. Given existing compliance with 33 CFR Parts 154 and 155, preauthorization plans do not have to restate all of these assets but should be able to refer to this regulated cap ability. With current U.S. stockpiles containing close to 1 million gallons of dispersant and international stockpiles containing over 1.5 million gallons (as established via the Global Dispersant Stockpile), the requirement to assess dispersant manufacturing capability during the initial response is unnecessary.

We recommend that the Agency revise this section of the regulation to not require specific information on stockpiles, spray equipment, personnel, training or manufacturing and acknowledge that the USCG and the BSEE are regulating these aspects of spill response.

V. <u>Section § 300.910(a)(2) – Preauthorization Plan Approval</u>

A. *The proposed language for w hen a withdrawal of concurrence invalidates a Preauthorization Plan is unclear.*

EPA needs to clarify when withdrawal of concurrence invalidates a preauthorization plan. In the preamble to this section (column 2, 80 Fed. Reg. 3389), the Agency states:

> *"The Agency is proposing specific procedures for concurrence withdrawals, allowing agencies to do so if they believe the preauthorization plan no longer addressed or reflects existing situations if it were to be implemented. While an agency with concurrence authority may not decide to withdraw concurrence from an approved preauthorization plan, there are curr ently no set procedures to promptly address those situations. The proposal would require the RRT and the ACs to address the withdrawal of approval of the preauthorization plan within 30 days of the withdrawal, allowing an opportunity to address the concerns."*

However, the Proposed Rule language is not clear regarding how the withdrawal process operates. While the proposed 30-day period for the RRT to address withdrawal is a positive step as it provides structure and guidance to the process, it is unclear w hen withdrawal of conc urrence would render the preauthorization plan invalid. API and NOIA believe this only makes sense if there is no active or ongoing response. Agencies should not withdraw their concurrence from an existing plan during a response as this could make responding in a timely fashion difficult, if not impossible. This could potentially allow an agency to invalidate a plan at the time of a s pill forcing a delay in the use of dispersants at the most critical time of the response. Note that the States of Alaska and Florida have withdrawn their approval for

preauthorization in state waters, which was done effectively without any additional regul atory requirements.

Clarifying the language to allow for preauthorization plans to remain in place dur ing the 30-day period in which RRTs and A rea Committees address the withdrawal would prevent disruption to response activities. This would maintain the validity and usefulness of preplanning and preauthorization, while allowing for agencies to withdraw con currence based on changing conditions, reducing the possibility of the withdrawal based on political or media influence. This 30 -day review period should also be subject to public comment.

We also note that currently Subpart J does not include the USCG representative to the RRT as a government agency required to approve preauthorization plans. It is important to inc lude one of the key stakeholders, the USCG, in the approval of preauthorization plans.

Accordingly, API and NOIA suggest modifying the proposed language as follows:

> *(2) Preauthorization Plan Approval. The EPA representative to the RRT, USCG RRT representative for the coastal a nd ocean zones where the USCG OSC would have primary authority and responsibility for operational decisions, the Department of Commerce and the Department of the Interior natural resource trustees and, as appropriate, the state RRT representatives shall review and either approve, approve with modification, or disapprove the preauthorization plans developed by the RRT and/or the Area Committees. Withdrawal of concurrence means the preauthorization plan becomes invalid and the authorization of use for chemical or biological agents must be performed according to paragraph (b) of this section. For an agency to withdraw concurren ce, the agency must first notify the RRTs and Area Committees. The RRTs and Area Committees shall review and address the withdrawal within 30 days of the date of notification, during which time the notification of withdrawal will be subject to public comment. The Preauthorization Plan will remain in effect while the RRTs and Area Committees review the withdrawal. If the agency withdrawing its concurrence remains unsatisfied after the 30 -day review period, then the Preauthorization Plan becomes invalid at th e conclusion of the 30 -day review period, and the authorization of use for chemical or biological agents must be performed according to paragraph (b) of this section.*

VI. Section § 300.910(a)(3) – Preauthorization Plans Reviews

A. *The re-approval process for pr eauthorization plans is too frequent and the timing and process for review and revision is unclear.*

Currently there are no requirements for the review of preauthorization plans. Revised Section 300.918(a)(3) requires preauthorization plan review, and revision as needed at the following times:

> *at least every 5 years; after a major discharge or a Spill of National Significance (SONS); to address revisions of the Schedule; to reflect new listings of threatened and/or endangered species; and to address any o ther change that may impact the conditions under which the use of chemical and biological agents is preauthorized.*

The preamble explains that "other changes" may include a new or revised worst case discharge estimate, and that this review requirement is "intended to ensure that preauthorization plans are actively maintained and updated to reflect revisions to the Schedule." (80 Fed. Reg. 3389). If the Agency decides to move forward with a re view requirement, API and NOIA recommend that the Agency clarify (1) what defines a

"major discharge"; (2) what process should be followed for re-approval; and (3) what effect a change in the preauthorization would have on existing FRPs and VRPs, which are also updated on a 5 -year cycle. We believe the review of preauthorization plans every 10 years or, at the discretion of the RRT/ Area Committee after a SONS (per [a] and [b] in the statement) , is sufficient and that the "major discharge" language should be removed . The Agency should also strike the requirement for review upon new threatened and/or endangered species listings and "to address any other change" since this type of planning on a "less than five year basis" overly -burdens the RRTs. Also, the Agency should clarify what the re-approval process would entail and establish a framework for how long review and approval should take. In addition, the USCG representative to the applicable RRT should also be included in this process. This, along with the remaining review triggers, would still allow for robust, current preauthorization plans, while maintaining stability of existing, potentially affected FRPs and VRPs.

We suggest modifying the proposed language as follows:

> *(3) Preauthorization Plans Reviews. The RRT and/or the Area Committees must review, and revise as needed, pursuant to paragraph (a)(1) of this section, preauthorization plans at least every 10 years; or at the discretion of the RRT and/or Area Committee after a Spill of National Significance (SONS); to address revisions of the Schedule; to reflect new listings of threatened and/or endangered species; and to address any other change that may impact the conditions under which the use of chemical and biological agents is preauthorized. The designated EPA RRT representative, the USCG RRT representative for the coastal and oceans zones where the USCG OSC would have primary authority and responsibility for operational decisions, the Department of Commerce and the Department of the Interior natural resource trustees, and the RRT representative from the state(s) with jurisdiction over the waters of the area to which a preauthorization plan applies shall review and either approve, approve with modification, or disapprove any revisions to the preauthorization plans.*

VII. Section § 300.910(b) Use of Agents Identified on the Schedule on Oil Discharges Not Addressed by a Preauthorization Plan

A. *A reasonable consultation time of 48 hours should be established for natural resource trustees if the Agency concludes that the current "if Practicable" standards should be changed*

In this section in the Proposed Rule (column 1 , 80 Fed. Reg. 3423), the Agency has removed the words *"when practicable"* with respect to consultation with the Department of Commerce (DOC) and the Department of the Interior (DOI) natural resource trustees for the following reasons discussed in the preamble:

> *"The requirement for consultation with the DOC and DOI natural resource trustees is also maintained. However, the language is amended by removing 'when practicable' with respect to consultation with the DOC and DOI natural resource trustees. The Agency believes that the case-by-case decision making should include consultations with natural resource trustees since these discharge situations may present unique challenges when selecting a response option that involves chemical or biological agents. While the Agency recognizes the time -critical nature of decision making during a response, advances in communication technology with increased capabilities to communicate quickly."*

(column 1, 80 Fed. Reg. 3390) API and NOIA believe that while technical advances in communication (e.g., cell phones) have improved since the last Subpart J revision, it does not guarantee an immediate or timely response by the required individuals. And the current system has worked well for years. Thus,

continued flexibility in the approval process (i.e., "when practicable") is still warranted to ensure the timeliest and best possible response . If the Agency concludes that the current " when practicable" language needs tightening up it could establish a reasonable consultation time of 36 hours. Spill response decision-making requires timely decision-making. Natural resource trustees already have an independent process to assess possible damages outside of the response effort (via Natur al Resources Damage Assessments [NRDA]). The phrase could be revised to state, "*...when practicable within a 36-hour timeframe*".

B. *The change in th e term "navigable waters threatened " will set up additional barriers to dispersant use and unduly limit actions by the OSC.*

In the same paragraph (column 1, 80 Fed. Reg. 3390), the Agency makes the following statement:

> *"The Agency is also proposing to revise the term 'navigable waters threatened' to 'waters and adjoining shorelines threatened' to be consistent with the provisions in paragraph (a) of this section."*

We are concerned that this change will set up additional barriers to dispersant use and unduly limit actions by the OSC.

C. *Early dispersant use decision-making should focus on documenting that a NEBA justification exists to support dispersant application.*

The Proposed Rule would unnecessarily establish new requirements f or an OSC to document the parameters used to authorize dispersants. This is discussed in the same section (column 2, 80 Fed. Reg. 3423), as follows:

> *"...the OSC must consider and document the parameters for the use of agents including the quantities to be used, the duration of use, the depth of water, the distance to shoreline and proximity to populated areas, and should address factors suc h as environmentally sensitive resources or restricted areas that might be impacted, agent inventory and storage locations, agent manufacturing capability, availability of equipment needed for agent use, availability of adequately trained operators and appropriate means to monitor agent use in the environment."*

Because dispersants have a limited window of opportunity to be used on spills, it is important to decide quickly on their use. Decisions by the OSC should not be delayed to complete information that is not needed. Many of the proposed items for the OSC to document (e.g., stockpiles, personnel, spray assets) are already specified and required in the applicable response plan requirements of 33 CFR Parts 154 and 155. Given the current stockpile availability in the U.S. and in the Global Dispersant Stockpile, dispersant availability will not be a concern for weeks or months. As suc h, we believe that the immediate actions by the OSC should include a review of the pre -authorization plan(s) and a rapid , incident-specific NEBA to support the selection of response options that may include dispersant application as a means to lessen or eliminate impacts to sensitive shoreline resources or to pr otect worker health and safety. In addition, we recommend that the Pr oposed Rule require referencing the USCG -classified dispersant OSRO or other OSRO for many of the items requested.

VIII. <u>Section § 300.910(c) Burning Agents</u>

A. *Ignition devices for in-situ burning are not burning agents.*

Under the current rule an OSC may author ize use of a burning agent on a case -by-case basis after concurrence of the EPA representative to the RRT, the state RRT representative and in consultation with the DOC and DOI natural resource trustees, when practicable. The Agency proposes to authorize the OSC to approve the use of a burning agent without the approval or consultation with other agencies to facilitate expedited use in recognition of their increasing importance in response to spills and that small quantities of burning agents are used and are quickly consumed by the resulting fire. API and NOIA agree and commend the Agency for this proposal. Further, we agree that ignition devices should be excluded from the definition of burning agents.

We believe that chemical herder agents are includ ed within the scope of the term burning agents. In the preamble to this section (column 3, 80 Fed. Reg. 3390), the Agency makes the following statement:

> *"Further, because of the nature of burning agents and the proposed revisions to the authorization of use for these products, the Agency continues to believe it is not necessary to require product submissions for burning agents."*

We believe this statement also applies to chemical herder agents. Please clarify if this is correct.

IX. Section § 300.910(d) Exception

A. *EPA needs to clarify the phrases "protection of human life" and "near inhabited areas."*

In this section in the Proposed Rule (column 2 , 80 Fed. Reg. 3423), the Agency makes reference to "the protection of human life ." We believe it is unclear whet her "*protection of human life* " is inclusive of the protection of worker health and safety. The Agency should amend this section to clearly include the protection of oil spill response worker safety within this concept. This same section also references spi lls "*near inhabited areas.* " This language is confusing since spills where dispersant use serves an environmental benefit will rarely (if ever) be near inhabited areas.

B. *A time limit on protecting human health is inappropriate.*

We believe the establishmen t of a 48 h our limit to the OSC's use of authority to use agents to protect human health is inappropriate. There should never be a limit to the OSC's ability to make decisions that are aimed at protec ting human health or human life, and the exception shoul d be permissible until the threat to human health has subsided. The current rule adequately allows for the OSC to approve use of dispersants in such cases and EPA has not justified that the proposed requirement is necessary or provides any benefit to existing requirements.

X. Section § 300.910(e) Prohibited Agents

A. *Banning nonylphenol (NP) and nonylphenol ethoxylates (NPE) is not necessary for dispersants.*

API and NOIA submit that NP and NPE components should not be banne d. In the preamble to this section (column 2, 80 Fed. Reg. 3391), the Agency makes the following statement:

> *"The Agency is also proposing to add a prohibition from listing on the Schedule and from authorizing use of any chemical or biological agents that contain n onylphenol (NP) or nonylphenol ethoxylates (NPEs) as components."*

While we appreciate the Agency's concern about NP and NPE components, an outright prohibition seems inconsistent with the practice of allowing NP and NPE ingredients in a variety of widely-used cleaning products in the U.S. After use, these products are routinely flushed and transported into wastewater discharges on a daily basis. Spill response dispersant use is extremely infrequent and only occurs when a significant threat to shoreline or near shore resources is present. Banning these components in oil spill dispersants while allowing their wide spread use in other commercial products seems contradictory and prejudicial. API and NOIA requests that the Agency explain why NP/NPE is banned in dispersant products, but not in many more commonly manufactured products.

B. *Endocrine disruption testing has little relevance and is not necessary.*

We submit that endocrine disruption testing should not be added. In the same section (column 1, 80 Fed. Reg. 3392), the Agency states:

> *"Because validated testing methods are still being developed the agency is not proposing any test requirements with respect to endocrine disruption for products to be listed on the Schedule."*

We believe that adding any such testing provides little relevance in the context of oil spill response decisions and that testing efforts are better directed at the use of standardized toxicity tests with aquatic organisms using endpoints of population relevance (i.e., mysid and *Menidia* tests proposed).

C. *Mechanical response clean-up methods have greater limitations in offshore conditions than dispersants.*

Mechanical response clean-up methods have many limitations. In the same section (column 2, 80 Fed. Reg. 3392), the Agency makes the following statement:

> *"There are chemical and biological agent alternatives to sinking agents and to agents containing NP or NPE; <u>as well as mechanical methods</u>* (Ed. emphasis added) *for responding to oil discharges, including those situations that pose extreme threats or are time critical."*

We believe the Agency must realize the limitations that mechanical response methods have in offshore conditions. Weather conditions will preclude booming and skimming operations in sea states above six feet wave height, and even in ideal sea state conditions, responding to offshore spills historically has resulted in 5-10% of spilled oil recovered by mechanical recovery methods. For this reason, it is critical that responders have reasonable access to chemical response methods, because of the substantially higher encounter rates (80% or more) that can be achieved by treating the spilled oil with dispersants.

XI. <u>Section § 300.910(f) Storage and Use of Agents</u>

A. *No additional requirements should be put in place for product shelf life.*

Currently, the manufacturer is required to provide information on recommended conditions of storage and use for each product at the time an application for listing a product under Subpart J is submitted to the Agency. With regard to shelf life, the Agency is requesting comments on whether more information should be submitted on issues related to shelf life. Specifically, in the preamble to this section (column 1, 80 Fed. Reg. 3393), the Agency makes the following statement:

> *"To alleviate concerns that applications are submitted that establish an extended or indefinite shelf life for a product, the Agency is requesting comments on whether any*

> *additional data or information requirements should be included for the product listing determinations specific to a product's shelf li fe, or whether alternative approaches, such as limiting the shelf life for product categories to a given timeframe, should be considered."*

API and NOIA believe that no additional requirements should be put in place for product shelf life and thus it shoul d not change the current practice of having the manufacturer provide information on recommended conditions of storage and use for each product. Ensuring product efficacy (regardless of age) is the only relevant end-point. For example, research has shown that properly maintained stockpiles of Corexit 9500 have remained effective for 25 years. Given that periodic effectiveness testing is proposed, there is no justification for also mandating a shelf life that could potentially result in costly replacement of stockpiles that remain viable and effective.

If the Rule dictates a dispersant shelf -life, this could caus e an undue financial burden. We believe that suppliers may set low shelf-lives to protect themselves from liability or to promote future recurring s ales, even though there are well-documented studies indicating that properly stored dispersant stockpiles have maintained their effectiveness for decades. A 2008 study by Clark, *et al.*,[30] evaluated U.S. stockpiles of Corexit 9527 and concluded that batches of this dispersant have remained viable for spill response for over 30 years. Establishing shelf -lives for products makes sense if periodic testing is not specified. However, since the Proposed Rule specifies re -testing of the dispersant stockpiles, there is no justification for also requiring a shelf -life. We submit that the requirement for a shelf -life should be removed from the Proposed Rule and that the Agency should instead provide more clarity on product re - testing for efficacy. Specifically, the requ irement to conduct periodic 5 -year tests of effectiveness and toxicity is unclear. The sampling requirement has not been delineated in the Proposed Rule, which could be interpreted, in the extreme case, that every single dispersant container would have to be re-sampled and tested to remain in an "approved" status.

The Rule should clearly state that representative samples of batches/lots are tested for efficacy and that the testing interval take into account the aforementioned studies that demonstrated tha t dispersant stockpiles maintain effectiveness for decades. Further, we submit that there is no scientific justification for re-testing toxicity.

B. *Central dispersant stockpiles are typically maintained by OSROs, not by Responsible Parties or product manufacturers.*

Under the Proposed Rule the OSC would only be able to authorize use if the Responsible Party certifies that the products have been stored under the conditions provided by the submitter. In this section in the Proposed Rule (column 2-3, 80 Fed. Reg. 3423), the Agency makes the following statement:

> *"The OSC may authorize for use only products that are certified by the Responsible Party to have been stored under the conditions provided by the submitter under 300.915(a)(6) and whose date of use does not exceed the expiration date listed on the container's label at the time of the incident. The Responsible Party must provide the OSC product documentation, developed in consultation with the submitter of the product to the Schedule, prior to OSC authori zation of product use affirming it has maintained its integrity, including no changes in its composition, efficacy, and toxicity."*

[30] Clark, J., Becker, K., & Lessard, R. (2008). Maintaining dispersant stockpiles and assessing their quality. *Proceedings of the 2008 International Oil Spill Conference, 2008*(1), 695-698. doi:10.7901/2169-3358-2008-1-695

The Agency needs to recognize that central dispersant stockpiles are typically maintained by OSROs and are stored mainly in 200 to 300 gallon totes so that they can be transported by truck or air to the site of a spill. In most spill situations, the dispersant stockpile will not be in the control of the Responsible Party or the submitter of the product (i.e., the original manufacturer) at the onset of a spill. As a result, there is no way for these two entities to make a certification for a product that was not in its direct control just prior to the spill. API and NOIA suggest modifying the language to clarify that Responsible Parties must only demonstrate compliance with Section 300.915(a)(6) for inventory/product they control.

C. *Certification of storage conditions should not be required for authorization of use.*

Section § 300.910(f) states that the OSC may authorize for use only products that have been stored under the conditions specified by the manufacturer including the maximum, minimum and optimum temperatures as well as humidity and other relevant conditions. This proposed regulation could require dispersants to be stored in climate controlled buildings with continuous temperature and humidity monitoring. We do not support this proposed requirement.

D. *API and NOIA opposes any re-testing requirements during a spill response.*

The intent of this section is difficult to discern but it appears that the certification requirement appears to essentially requires complete retesting of the product (at the time of the spill) for the Responsible Party to certify such a statement. It appears that the Agency has included this language with the specific intent to delay the dispersant authorization process by requiring complete re-testing at the time of a spill incident. We strongly oppose any re-testing requirements during a spill response. Any product testing should be conducted at a specified, planned interval, not during an emergency response. During a response, the Responsible Party should not have to certify the dispersants that will be applied, since that is the purpose of the NCP Product List. The only requirement should be for the Responsible Party to show the OSC, if requested, the original manufacturer's verification (Certification of Analysis) that the product meets specifications for dispersants with an age of less than the period for retesting recommended by the manufacturer. For older dispersants the Responsible Party should, if requested, show the efficacy re-test data for the stockpile. The test required should only be for effectiveness as it is a reliable indicator that the composition has remained the same. We propose that this language is changed to read: *"The OSC, to authorize the use of dispersants, may request that the Responsible Party provide test data of the dispersant stockpile showing it is effective."* We also suggest modifying the language so that it applies only for product manufactured or produced after the date of final rule publication.

E. *Re-testing requirements must permit flexibility in test results over time.*

This same section specifies re-testing and re-labelling requirements. This section does not specify the criteria for re-testing results and needs to specify permissible levels of deviation from past testing results. We point out that efficacy and toxicity testing have inherent variability. Two testing runs of the same product, in the same laboratory, in the same week can yield varied results, so any re-testing requirements must permit flexibility in test results over time. We suggest an allowable 10% variance in effectiveness test results, based on manufacturer and scientific experts' opinion on effectiveness testing. As previously stated, we do not believe there is any scientific justification to re-test for toxicity since efficacy results will confirm if the composition of the dispersant has changed. We suggest clarifying language so that it is clear product testing is only required if it is not stored under requirements provided by submitter, or it has expired.

F. *The assignment of an arbitrary 5-year life to products is biased and creates unwarranted expense due to recurrent testing.*

Finally, the assignment of an arbitrary 5-year life to products without regard to prior research on product viability over time, or on manufacturer input on product re-testing, is biased and creates unwarranted expense due to recurrent testing. Proposing 'no change' in composition, efficacy, and toxicity seems unreasonable and could result in significant costs of replacing usable dispersant stockpiles. Additionally, dispersant manufacturers typically advise that container seals not be broken and the material not be exposed to air, which would be necessary to test existing stocks, which may or may not have an expiration date printed on the label under current regulation.

Studies exist that show Corexit products maintain effectiveness for 25 years. We believe that any re-certification term should be based on existing scientific and manufacturer data. There should also be an allowance for some change/degradation of product over time, aligned to international practice. For example, the United Kingdom Marine Management Organisation allows a loss of efficacy to 75% of the original minimum level. Further, there is no scientific rationale to require any form of toxicity re-testing of a dispersant that was previously tested. Manufacturers will have the best insight regarding when product recertification should be needed. In addition, the Responsible Party should never be required to re-label products they did not produce, with the possible exception of adding a re-test date for product efficacy.

XII. Section § 300.910(g) Supplemental Testing, Monitoring, and Information

A. *API believes the standard efficacy and toxicity tests already required are more than adequate for regional use.*

We do not believe there is a need for additional testing than is currently required under the NCP. In the preamble to this section (column 2, 80 Fed. Reg. 3393), the Agency makes the following statement:

> *"The proposal removes 'in addition to the test methods specified in 300.915 and described in appendix C to part 300.' While RRT's may want to use the efficacy and toxicity testing protocols specified in Appendix C of the NCP for comparative purposes, the proposed revisions clearly establish the RRT's authority to require tests using parameters beyond those specified in Appendix C. Furthermore, there may be supplementary toxicity and efficacy testing information based on recognized standard testing methods already available that RRTs may want to consider when addressing site, area, or ecosystem specific concerns."*

We believe the standard efficacy and toxicity tests already required are more than adequate for regional use. We suggest the Agency consult the NOAA Dtox database. A review of that database suggests that the sensitivities of all tested species are in the known range and that additional tests simply reinforce the existing species sensitivity distribution curves. Regional data are unlikely to fundamentally change overall NEBA considerations for dispersant use. Any regionally required laboratory efficacy tests during a response will not be as relevant as a pilot test in the field and will only delay the dispersant authorization process approval beyond the window during which dispersants could be applied (thereby precluding their use in some situations where they may have provided a net environmental benefit). This proposal allows RRTs to request a broad range of additional data on a product, a situation that creates the potential for confusing and costly testing and is unlikely to add further information of value to oil spill response decision-making.

B. *Additional monitoring during a response is unreasonable.*

Section § 300.910(g) provides the RRT with the authority to request that the OSC require the Responsible Party to conduct additional monitoring associated with use of a product. We submit it is unreasonable to require additional monitoring during a response except in exceptional circumstances. In the preamble (column 2, 80 Fed. Reg. 3393), the Agency makes the following statement:

> *"For example, the RRT may want to monitor the exposure of marine mammals to oil constituents, including the dispersed oil, or to monitor toxicity in the water column using biological assays."*

We believe such monitoring during a response is unreasonable and should only be undertaken if (a) there is some critical question associated with the spill response that has not been answered with the testing already in place; and (b) there is sufficient time to arrange for proper and scientifically defensible protocols to be established. Given that tests associated with such monitoring are likely to be complex and require considerable time to prepare, implement, and interpret, it appears that the tests would not inform operational decision-making during the response, but rather would develop data for NRDA. As previously stated, operational monitoring and NRDA are two different things and this Proposed Rule should not specify NRDA monitoring. Additionally, the "biological assays" that are referenced in the Proposed Rule are impractical given the nature of offshore spills where dilution is significant, water movement is constant, and synoptic sampling of small quantities of organisms from dynamic water masses is not representative of any chronic or resident conditions.

API and NOIA suggest revising the language as follows:

> *(g) Supplemental Testing, Monitoring and Information. The RRT may require supplementary toxicity and efficacy testing, or available data or information that addresses site, area, or ecosystem specific concerns relative to the use of a product for both planning and authorization of use. During a discharge incident, the RRT may request that the OSC require a Responsible Party to conduct additional monitoring associated with the use of a product to the extent necessary for conducting response operations. Such additional monitoring data may include supplemental sampling and testing to ensure safety of response personnel, to assess acute environmental impacts or otherwise aid the OSC and/or the RRT in operational decisions.*

XIII. Section § 300.910(h) Recovery of Agents from the Environment

A. *The OSC has the authority to use the NEBA process to make decisions on recovery of agents from the environment.*

In the preamble to this section, the Agency is requesting comments on new language that would require the Responsible Party, under OSC oversight, to remove specified products/agents from the environment after use. While the Proposed Rule hints at NEBA, it does not reference it specifically when suggesting that some situations (e.g., safety of personnel or greater harm to the environment) might preclude the removal of these agents. That would seem to contradict the main change in this section (h) while not providing any framework (e.g., NEBA) in which to make a determination for precluding the removal. The language is also potentially redundant as the OSC is already obligated to consider safety and additional adverse environmental impacts (i.e., via NEBA) as part of the normal decision-making process. Essentially, it is already an underlying tenet of the response process. Additionally, in some cases recovering 100 percent of these agents after application is not practical or even feasible regardless of the diligence of the Responsible Party

We suggest revising the proposed language to make it clear that the OSC has the authority to use the NEBA process to make these decisions:

> *(h) Recovery of Agents from the Environment. Depending on factors such as the safety of response personnel and harm to the environment (including consideration of Net Environmental Benefit Analysis [NEBA]), and as directed by the OSC, the Responsible*

Party shall, following application, endeavor to adequately contain, collect, store and dispose of the agents that are intended to be recovered from the environment.

XIV. Section § 300.910(i) Reporting of Agent Use

A. *The Agency should not adopt a prescriptive time period for reporting of agent use.*

The Agency is requesting comments on the appropriateness of requiring the OSC to provide the RRT additional reporting information " *within 30 days of completion of agent use* ." This runs counter to the adaptive decision-making that might be required during a discharge event where such agents are used.

Based on the pre authorization testing that is being proposed and, more importantly, the variability of discharge situations, we recommend that the Agency not adopt a prescriptive time period and rather provide the relevant RRT(s) discretion to determine the appropriate time period for reporting agent use. A strong justification for this approach is that during prolonged discharges, a fixed 30 -day reporting period might constrain the response personnel and interfere with higher priority resource allocations. Furthermore, the OSC would typically request this information as part of the daily briefing.

API and NOIA suggest removing the 30-day window for reporting and modifying the proposed language as follows.

> *(i) Reporting of Agent Use. Unless already included in the OSC report required under § 300.165 of this part, the authorizing OSC shall provide the RRT the following information on chemical and biological agents used in response to an oil discharge: product name, quantity and concentration used, duration of use, and locations.*

XV. Section § 300.913 Monitoring the Use of Dispersants

A. *The appropriate use of spill mitigating products will reduce the overall impact of the spill and protect the health and safety of response personnel.*

The Agency makes the following two related statements to explain the goal of the NCP Product Schedule (column 3, 80 Fed. Reg. 3393; column 3, 80 Fed. Reg. 3402; respectively):

> "*The goal of establishing a Schedule under the NCP is to protect the environment from possible damage related to spill mitigating products used in response to oil discharges.*"
>
> *"Emergency response personnel need to know whether a dispersant or any other type of chemical or biological agent on the Schedule could have negative environmental impacts relative to the oil before decisions are made about its use in a particular oil discharge situation. Consequently, it is essential to consider comparative information about the efficacy and the toxicity of these products.*"

Possible damage related to spill mitigating products used in response to oil discharges is considered, but not the possible damage that could be caused by the oil itself without the use of spill mitigating products. The main threat to response personnel, communities, and the ecosystem comes from the oil, not from the spill mitigating products. The concept of CERA/NEBA accepts that some effects might be caused by the response (including the use of spill mitigating products), but also recognizes the greater benefit of reducing the overall impact of the spill that can be achieved by using them. The Agency needs to acknowledge the importance of NEBA in justifying the use of dispersants.

We believe that the focus on the potential for negative environmental impacts that could be caused by a dispersant ignores the essential point that dispersants are used to mitigate (minimize) the negative human health and environmental impacts that could be caused by the oil. The main threat to the ecosystem comes from the oil itself, not from the dispersant. The wording of the preamble to a final rule needs to:

- reflect the ultimate aim of dispersant use;
- clarify that the toxicity of dispersed oil comes from the oil (not the dispersant); and
- acknowledge and emphasize the importance of using a NEBA process in dispersant use decision-making.

We request that the Agency consider including the following two paragraphs (or equivalent language) in the preamble to the final rule to briefly explain dispersant use and the importance of NEBA in making dispersant use decisions.

> *"Dispersant Use and Net Environmental Benefit Analysis (NEBA)*
> *The aim of dispersant use is to protect response personnel from VOC exposure and to prevent oil from drifting ashore and potentially causing severe and long-term damage to coastal resources and communities, by dispersing the oil into the water column where it will be diluted rapidly to very low concentrations and the majority of the oil will subsequently be biodegraded. This common aim of dispersant use is the same, whether it is applied to oil on the sea surface or directly at the outflow source in the case of a subsea release. The potential risk of dispersant use is that, by dispersing the oil into the water column, there is an increased potential for toxic effects to be caused to marine organisms. The severity of the toxic effects that could be caused is related to the exposure of the marine organisms to the dispersed oil (or the partly water-soluble oil compounds) and exposure is a function of concentration of the oil in the water and the duration for which such exposure persists.*
>
> *The issue of the environmental trade-offs caused by dispersant use is addressed by Net Environmental Benefit Analysis (NEBA). NEBA is a process used to make choices about oil spill response techniques to be used to minimize the potential impacts of the oil on people and the environment. NEBA takes into account the likely effectiveness and feasibility of conducting alternative response techniques under the prevailing conditions. The main principle of NEBA is that an oil spill response technique (or combination of techniques) that is used should be that which minimizes potential impacts of the oil on the environment to the greatest extent and promotes the most rapid recovery and restoration of the affected area. In many cases of dispersant use, the potential risks to the offshore marine ecosystem are very low because dispersed oil concentrations are low and persist only briefly. There is benefit in using dispersants in offshore environments to prevent damage to coastal communities and sensitive near-shore resources where persistent stranded oil can cause longer-term environmental problems."*

The Proposed Rule also appears to focus more on the environmental effects during the response rather than the health and safety of response personnel. The Agency should make it clear that one of the primary objectives of characterizing the efficacy and understanding transport is to protect the health and safety of response personnel. That critical aspect is not mentioned in this section.

B. *The OSC should determine where, when, and how monitoring resources should be deployed.*

EPA should clarify the meaning of the phrase *"upon initiation and for the duration of subsurface dispersant use."* Later in this section of the preamble (column 2, 80 Fed. Reg. 3394), the Agency states the following:

> *"The proposal requires the responsible party to implement monitoring for any subsurface dispersant use in response to an oil discharge upon initiation and for the duration of subsurface dispersant use."*

The phrase *"upon initiation and for the duration of subsurface dispersant use"* can be misconstrued to mean that all monitoring required by the Proposed Rule must be performed at all times, when in fact the monitoring timing and operational parameters should be established by the OSC. Every response is unique, and it should be left to the OSC to determine where, when and how monitoring resources should be deployed to provide the best near real-time operational monitoring on the progress of the response.

C. *The Proposed Rule should be restructured to separate surface dispersant application monitoring from subsea dispersant injection monitoring.*

EPA should distinguish between surface and subsea dispersant application monitoring. In the first paragraph of this section in the Proposed Rule (column 1, 80 Fed. Reg. 3424), the Agency states:

> *"As directed by the OSC, the Responsible Party must monitor any subsurface use of dispersant in response to an oil discharge, surface use of a dispersant in response to oil discharges of more than 100,000 US gallons occurring within 24 hours, and surface use of dispersant for more than 96 hours in response to an oil discharge, and submit a Quality Assurance Project Plan for approval to the OSC covering the collection of all environmental data".*

We believe it is significant that monitoring is to be required *"as directed by the OSC."* It should be made clear that the OSC has discretion to authorize initial use of dispersants during a time-limited efficacy testing phase while evaluating the need for and extent of additional adaptive monitoring actions. The OSC is in the best position to determine when and what operational spill response monitoring is to occur. Experience has shown that the use of phased monitoring approaches, consistent with the USCG *et al.*'s *Special Monitoring of Applied Response Technologies* (SMART) protocols, is appropriate, and they are most responsive to changes in site-specific conditions and information requirements. While the requirement for the Quality Assurance Project Plan (QAPP) is constructive in helping to maintain effective data integrity and management, the rule should provide the OSC with discretion in how detailed the document needs to be to meet this requirement. This would provide the relevant incident decision-makers with practical information that can be consistently acquired, analyzed, and understood.

The Proposed Rule seems to inappropriately replace the currently used 3-tier SMART protocols for monitoring surface dispersant use with the more exhaustive monitoring approach proposed for subsea dispersant use. The USCG Strike Teams already have the responsibility for monitoring surface dispersant operations and possess the trained personnel and equipment to conduct these activities. Many RRTs require the immediate activation of USCG Strike Team monitoring for all dispersant operations, regardless of size or location. We believe it is beneficial to have the government perform the surface monitoring to enhance credibility with the public that the operations are being conducted correctly and are working as intended. If this practice is going to change, the Agency should explain why it is circumventing an established, well-vetted process performed by the USCG Strike Teams. The information in the Proposed Rule creates confusion between existing responsibilities of the USCG Strike Teams and those of the Responsible Party. We request that this confusion be reconciled between the Agency and the USCG, as the Responsible Party should not be required to provide duplicate monitoring for surface dispersant operations.

For these reason s, we believe this section of the Proposed Rule should be restru ctured to separate surface dispersant application monitoring from subsea dispersant injection monitoring. API and NOIA offer the following relevant points:

- The section on surface dispersant application monitoring should specify that it will be implemented with existing SMART protocols using USCG resources.
- The section describing subsea dispersant injection monitoring should specify it will be performed by the Responsible Party.
- In both cases, the Proposed Rule should clearly state that the mobilization o f monitoring capabilities will not delay initial dispersant applications. Monitoring requirements should not impede the use of chemical agents during a response and should be implemented as appropriate resources become available during the response.
- The c riteria of greater than 100,000 gallons or duration of 96 hours for the spill should be replaced with a single criterion that " *any enhanced monitoring beyond SMART shall commence within seven days.* " Our rationale for a deployment schedule of seven days for any enhanced monitoring (whether subsea or surface dispersant monitoring) ensures that scientifically - defensible monitoring can be performed, by ensuring:
  - The best scientific experts can be assembled and mobilized to the spill;
  - Once on scene, the science team can design a monitoring plan that can be integrated safely into ongoing response operations, and be worked into the Simultaneous Operations (SIMOPs) schedule;
  - The team can tailor a general monitoring strategy into a specific mission plan for the monitoring vessel that is available (there is no way to know what research vessel may be available, and each vessel will have unique characteristics for equipment deployment);
  - Available monitoring vessel(s) can be identified, mobilized to port, and properly out fitted with the scientific equipment needed to implement the mission plan;
  - An appropriate sampling strategy can be developed and approved for the specific incident, taking into account ongoing response activities that may impede access to sampling sites; and,
  - Appropriate safety considerations for the monitoring vessel, dispersant application or injection platforms, and rapidly-assembled teams can be properly reviewed.

D. *Effectiveness determination is not currently scientifically feasible.*

The language in this section of the Proposed Rule appears to indicate the Agency believes that comprehensive monitoring will permit the dispersant effectiveness at sea to be quantitatively determined. This requirement for effectiveness determination is not currently scien tifically feasible. Also, use of the word "*comprehensive*" should be avoided as it is subjective. Instead, it should be replaced with "*adaptive*" which indicates that a decision could be revisited as the d ischarge situation changes. We recommend replacing all instances of the word " *comprehensive monitoring*" with "*adaptive monitoring*" throughout the proposed section. The following language could be used:

> *The agency believes that an adaptive monitoring program which includes objective science and ocean transp ort modeling in certain discharge situations is necessary to determine the overall effectiveness of the dispersants and should transcend the initial dispersant application to include the extent, character, and transport of the dispersant and dispersed oil in the water column.*

With respect to <u>surface dispersant use monitoring</u>, we offer the following comments:

- Experience at the DWH and other incidents has shown that no currently available remote sensing technique, including airborne visible/infrared spectro meter (AVIRIS) and other multi- and hyper-spectral equipment, is capable of accurately quantifying the total amount of oil on the sea surface. It is also not possible to accurately determine the total amount of dispersed oil in the water column at any time because water sampling or ultra violet fluorometry (UVF) transects cannot be made at sufficient resolution in space (i.e., distance and depth) and cannot be made at sufficient resolution in time (i.e., simultaneously at all points in the water under an oi l slick). Localized measurements by UVF made before and after dispersant spraying can reveal the relative effect of dispersant use, but is not quantitative . A sharp increase in dispersed oil concentrations in the water, after dispersant use, is a good indi cation that the dispersant is working, but such measurements cannot produce an aggregate estimate of disper sant effectiveness. As such, we believe that the Agency should remove any requirement for quantitative reporting of efficacy, since it is not technically possible.
- A QAPP should not be required for surface dispersant operations. QAPP for approval by the OSC covers the collection of all environmental data to ensure and maximize its quality, objectivity, utility, and integrity. All these requirements wil l result in the need to generate a very large document that needs to be reviewed, verified and approved by OSC. This will lead to delays in dispersants decision-making, while adding little value to the goals of the response. A more useful approach may incl ude a daily Quality Control review of the SMART data to determine if the surface dispersant applications were effective that day, to support decisions for possible dispersant use the next day. A Quality Control form was developed during DWH for this purpos e. we believe the QAPP requirement should be deleted for surface dispersant operations and replaced with a simplified Quality Control form for SMART data.

With respect to <u>subsea dispersant injection monitoring</u>, we provide the following comments:

- The Age ncy does not address methods for visual determination of subsea dispersant effectiveness. During DWH, visual monitoring of the subsea dispersant injection was conducted in two ways to provide valuable, near real -time information to help inform response dec isions. These included:
  - Aerial observations of surfacing oil quantity and appearance (provided from observation aircraft overflights); and
  - Subsea observations of changes in plume appearance at the injection site (provided from ROV imagery).

  This form of monitoring for subsea dispersant injection should be recognized as an important step for initially evaluating the operation while more "enhanced scientific monitoring capabilities" are being assembled.
- API has developed subsea monitoring guidance (API Techn ical Report 1152 , *Industry Recommended Subsea Dispersant Monitoring Plan – Version 1.0*) and recommends the Agency consider this as the basis for their approach for operational monitoring. This guidance is publicly available at http://www.oilspillprevention.org/~/media/oil-spill-prevention/spillprevention/r-and-d/dispersants/api-1152-industry-recommended-subsea-dis.pdf.

XVI. Section § 300.913(a) Documenting and Monitoring the Oil Discharged and Dispersants Used Requirements

A. *The Responsible Party should not be required to document information at the beginning of a response that has already been determin ed in planning and preparedness activities.*

The current regulations found in 33 CFR Parts 154 and 155 require OSROs that use dispersants to maintain a certain level of spray assets, dispersant stockpile quantities, and logistical capabilities to

deliver stockpiles to staging bases, and to provide information on the application dosages they can apply. Since this is regulated elsewhere, reporting of these elements should not be required in the Proposed Rule and should therefore be deleted.

The Responsible Party should not be required to provide the *"rationale for dispersant product choice(s) including the results of any efficacy and toxicity tests specific to area or site conditions."* If the dispersant product is on the Product Schedule (which means that eff icacy and toxicity data have already been provided to the government), then it is, by definition, a viable option for the response.

Providing *"hourly application rates"* does not apply to aerial and vessel application since dosages are prepared in gallons per acre application and are based on the spray assets' application speed an d spray system swath width. We suggest that this statement be modified to read: *"hourly application rates (in the case of subsea dispersant injection)."*

The phrase " *recommended di spersant-to-oil ratio (DOR)* " should be modified to " *recommended <u>initial</u> dispersant-to-oil ratio (DOR).* " Every spill is different, and the DOR should be adjusted as additional monitoring, analytical data results, and/or oil release rate information becomes available. Additionally, the DOR may need to change as the oil weathers or other conditions change. The Agency should consider providing a clarifying sentence recognizing that oil characteristics and effective DORs may vary and thus dispersant application rates/volumes cannot be fixed to pre-determined values as this practice may affect negatively the response.

The Agency should refrain from requiring estimates of dispersant volumes needed to effectively respond to the spill at the onset of the response. In the case of a subsea oil release, it is impossible to know how much oil will be released and , therefore, impossible to estimate what volume of dispersants might be necessary. Further, as oil weathers, DORs for surface application will need to be adjust ed, and these changing weathering conditions cannot be fully known in advance. Given current dispersant stockpiles in the U.S. and in the Global Dispersant Stockpile, it should be assumed that sufficient dispersants will be available if their use is warranted by NEBA.

B. *Identifying and reporting associated volatile petroleum hydrocarbons will be impossible with changing environmental conditions, and is unnecessary for response decision-making.*

The requirement to *"...document any associated volatile petroleum hydrocarbons..."* would be very difficult to achieve, given that volatiles will fluctuate with sun exposure and changing wind conditions. The reason for this requirement is not explained; therefore, the Agency should specify what is intended by this requirement or delete it.

C. *The Agency should accept best available industry practice for dispersant monitoring.*

There is no need to restate the requirement for documenting the results of toxicity testing. In general, the list of characteristics is practical and do es not create an undue burden ; however, there is a significant assumption in the requirements (i.e., that the condition of the discharge point is such that it will enable the acquisition of these data at any given time). A reference to a best available industry practice, or any of the existing guidance on dispersant sampling/monitoring (e.g., API guidelines on subsurface dispersant monitoring, API TR 1152) would be more effective for gathering the information requested by the agency without imposing undue b urden on response personnel. Additionally, there is no technical reason to differentiate subsurface discharges from the rest of this paragraph. We suggest removing this differentiation and suggests modifying the proposed language as follows:

As directed b y the OSC, the Responsible Party must monitor any subsurface use of dispersant in response to an oil discharge, surface use of dispersant in response to oil discharges of more than 100,000 U.S. gallons occurring within 24 hours, and surface use of dispersa nt for more than 96 hours in response to an oil discharge, and submit a Quality Assurance Project Plan for approval to the OSC covering the collection of operational monitoring data. When these dispersant use conditions are met, and for the duration of dispersant operations, the Responsible Party shall:

(a) Document the characteristics of the source oil; best estimate of the oil discharge flow rate, periodically reevaluated as conditions dictate, including a description of the method, associated uncertainti es, and materials; dispersant(s) product used, rationale for dispersant product choice(s), recommended dispersant-to-oil ratio (DOR); and the application method and procedures, including a description of the equipment to be used, hourly application rates, capacities, and total amount of dispersant needed.

XVII. Section § 300.913(b) Sampling Requirements

A. *Daily water sampling and testing is burdensome.*

The requirement to conduct these required tests on water samples daily may prove overly burdensome and not necessary after the initial efficacy tests. We suggest modifying the Proposed Rule to ensure that the OSC has discretion in h ow frequently the samples must be acquired after the initial efficacy tests are conducted. Sampling on a daily basis, once efficacy is determined and agreed, is arbitrary and does not fit the timeline required to provide in-depth analysis of samples.

The collection of water column samples where the dispersed oil plume is located is not necessary and in many cases is not possible for aer ial dispersant application. For spills of up to 100,000 gallons, aerial dispersant assets can be activated and apply dispersants to the entire spill area during Day 1, before a monitoring team can arrive on-site to conduct water sampling. It was shown during the DWH event that to collect water samples, special operations had to be developed and coordinated so that the vessel monitoring team had sufficient time to obtain the water samples of the dispersed oil. This was due to the fact that most of the area u nder the aerial dispersant application returned to normal background levels within one hour following the application. It should also be understood that water samples will only determine the amount of oil in the sample, not the effectiveness of the dispers ant, as there is no way of determining the amount of oil that was treated. As stated previously, these sampling requirements do not provide information on dispersant effectiveness and can therefore not inform the operational response. Water sampling requir ements are more appropriate to NRDA and should be directed under NOAA jurisdiction.

B. *The oil droplet size distribution measurement, as described, is not technically feasible.*

The requirement in 300.913(b)(1) to conduct *"in situ oil droplet size distribut ion, including the mass or volume mean diameters between droplet sizes ranging from 2.5 to 2000 µm* " is not possible for three reasons:

- The General Information section suggests that this measurement can be done via a Laser In Situ Scattering and Transmissometry (LISST) instrument, but the LISST detection limit is roughly 500 microns;
- Commercial droplet size measurement equipment suitable for deep water monitoring that ranges up to 2000 microns does not exist; and

- Even if such instrumentation became availab le in the future, there would be no way to safely deploy it in the resulting dispersed oil plume by either ROV or research vessel without interfering with other critical, oil spill response operations.

With respect to droplet size determination, the Agenc y should focus on the most important aspect of subsea dispersant injection, which is to significantly reduce the initial droplet size to slow and change the surface expression of oil release as well as "thin out" the surface slick for better dispersion on the surface. Work by API and sponsored by BSEE suggest that HD high speed photography or sonar are the most promising methods of studying changes in droplet size during a full scale applic ation of subsea dispersants. We propose that alternative methods (i. e., high definition, high speed photography; sonar; etc.), as they mature and become available, should be allowed. Additionally, there should be flexibility to use other, improved scientific methods as they become available.

C. *Fluorometry is only useful as a screening tool.*

Indirect measures, such as fluorometry, are useful only as a screening tool to help refine monitoring locations and not as direct evidence of oil or specific oil concentrations in the subsurface as suggested.

In 300.913(b)(2), in addition to *in-situ* fluorometry, the Proposed Rule also requires the Responsible Party to conduct a fluorescence intensity analysis on collected water samples to determine fluorescence signatures of the dispersed oil for use as a relatively simple and rapid mea ns to assess dispersion effectiveness. Fluorescence intensity in this context is redundant and not as meaningful a measure as particle size analysis. An alternative would be to simply suggest this as a potential measure that could be used as the situation demands. Moreover, dispersant effectiveness assessments using SMART Tier 1 visual monitoring is a relatively simple, rapid way to evaluate this and is a well-accepted practice, as is the use of simple fluorometry from the SMART Tier II protocol for surface dispersant application. The Agency should substantiate the need to replace the tiered monitoring approach of the SMART protocols and, rather than listing all possible tests that can be conducted, specify how they add value to response decision-making and offer improvements over other, existing approaches.

D. *Detailed chemical analysis is unnecessary and many of the tests are too di fficult to perform in the field.*

In 300.913(b)(4), the Agency should recognize that more real-time operational monitoring information can be obtained by "quick screening" some samples for total petroleum hydrocarbons (TPH) by means of a hand-held gas chromatograph flame ionization detector (GC -FID). Detailed chemical analysis will not inform the response decision-making process and should instead fall under the purview of NRDA.

The requirement in 300.913(b)(5) for carbon dioxide measurements is unclear. While pH and dissolved oxygen monitoring seem logical, carbon dioxide (CO_2) measurements are not routinely made in deep water. It is not clear from the Proposed Rule what benefit carbon dioxide monitoring would bring that dissolved oxygen does not already provide. Similarly, the proposed requirement to measure methane is also unnecessary as it is also linked to potential oxygen deple tion, and the measure of DO is a reliable and sufficient method. There is a limited amount of sensors and water sampling devices that can be deployed from a research vessel at one time and inclusion of CO_2 detection complicates other elements for the deployment of the instrument array.

Regarding the DO measurements, we suggest simply requiring the *in situ* measurement of DO using best available technology (which would currently be automated sensors) to measure dissolved oxygen at depth and not expose it to surface pressures and temperatures before or during analysis. Verification can be achieved through the use of consistent sensor cleaning procedures, calibration tests, and redundant sensors which can be compared. Winkler titration, as discussed in the Proposed Rule, is an outdated

analytical method that is impractical for use on unstable offshore vessels, is prone to the introduction of human errors, and would slow the information flow within the response. If periodic bench tests are required to verify the integrity of the *in situ* DO sensor measurements, there are more consistent and reliable techniques than Winkler titration. We also recommend that only a fraction of the water samples be used to verify the *in situ* DO measurements as opposed to every sample , which will slow the process and information flow.

With respect to TPH analysis, the recommendation would be to provide these data for a fraction of the water column samples as opposed to every water sample (as currently proposed).

The requirement in 300.913(b)(7) for analyzing heavy metals has no bearing on dispersed oil monitoring. There is no operational reason for collecting these data. This should fall under the purview of NRDA. The source samples of oil are much better sources for determining the proportion of individual constituents. The carbon dioxide, methane, and heavy metals analysis requirements do little to inform response decisions, add unnecessary time and cost to the response activities, and would have a high degree of uncertainty in the absence of understanding the natural variations that might occur in the water column or in the case that a compounding discharge event (e.g., major flood) occurs during the response.

The focus of the Proposed Rule is almost exclusively on empirical data a nd should include some elements of 3-D and 4-D modelling to broaden the overall understanding of subsurface conditions.

We suggest revising the proposed language as follows:

> *(b) In areas not affected by the discharge of oil, collect a representative set of background water column samples following standard operating and quality assurance procedures, at the closest safe distance from the discharge as determined by the OSC and in any direction of likely transport considering surface and subsurface currents and oil properties for the variables listed below. Adherence to API TR1152 meets the requirements of this subsection. In the dispersed oil plume, at an interval designated by the OSC, collect water column samples following standard operating and quality assurance procedures, at such depths and locations where dispersed oil is likely to be present to analyze for the following:*
>
> > *(1) In-situ oil droplet size distribution, including mass or volume mean diameter for droplet sizes, with the majority of data coll ected between the 2.5 and 100 µm size;*
> > *(2) In -situ fluorometry signatures targeted to the type of oil discharged and referenced against the source oil;*
> > *(3) Dissolved oxygen (DO);*
> > *(4) Total petroleum hydrocarbons (based on an extracted water sample), individual resolvable constituents including volatile organic compounds, aliphatic hydrocarbons, monocyclic, polycyclic, and other aromatic hydrocarbons including alkylated homologs, and hopane and sterane biomarker compounds;*
> > *(5) 3-D and/or 4-D modeling when/where practical*
> > *(6) Turbidity;*
> > *(7) Water temperature;*
> > *(8) pH; and*
> > *(9) Conductivity.*
>
> *The purpose of analyzing these components is to map the extent and location of subsurface oil ; to characterize the condition of that oil ; and to help inform transport models that are also a key part of the decision-making process by helping to characterize the overall movement of subsurface oil.*

XVIII. Section § 300.913(c) Dispersant Effectiveness and Oil Distribution Characterization

A. *The Proposed Rule needs to be clearer on what is required for surface monitoring and what is required for subsea monitoring.*

Section § 300.913 provides monitoring requirements collectively for "subsurface use of dispersant in response to an oil discharge, surface use of dispersant in response to oil discharges of more than 100,000 U.S. gallons occurring within 24 hours, and surface use of dispersant for more than 96 hours in response to an oil discharge…" However, the preamble is often unclear as to whether every requirement within this section applies to surface or subsea dispersant applications or both. To facilitate a better evaluation of the proposed changes, each subsection should be divided into those aspects that apply to subsea applications and those to surface applications and those that apply to both. Elements within this section are already addressed in other sections of this Proposed Rule, particularly water column monitoring. The sections ought to be combined. The concept of an informed monitoring program (one that includes fixed and adaptive sampling locations and modeling) is already well understood and acknowledged in the spill response community.

In 300.913(c), requiring best available technologies may result in equipment not suitable for field conditions. Consideration should be given to changing "*available*" to "*practicable*."

In 300.913(c), the best available technology requirement should acknowledge aerial photography as a tool to measure effectiveness. This was a key method of effectiveness assessment during DWH and is not mentioned in the Proposed Rule. It is much more practical than the concepts put forward in the present proposal.

The relative effectiveness of the surface application should be determined by using the SMART protocols. For surface application, it is impossible to determine a numerical effectiveness as the amount of oil on the surface that is being sprayed is not known. Note that the analysis equipment often cannot return to the spray site in time to capture the information requested as the dispersant plume quickly dilutes or cannot be found.

XIX. Section § 300.913(d) Ecological Receptor Characterization

A. *Monitoring required to determine possible environmental effects is too time consuming to support dispersant operations decisions.*

Industry agrees that environmental monitoring should be performed to support operational decision-making. Monitoring required to determine environmental effects, however, is likely to be too time consuming to support dispersant operations decisions and should remain a NRDA activity.

This section of the Proposed Rule requires the Responsible Party to characterize the ecological receptors and to consult with the OSC to determine an acute toxicity level of concern for the dispersed oil using available dose/response information relevant to potentially exposed species. Conducting the required ecological characterization of the spill site may not be possible in the available response time-frame. If the untreated oil is likely to drift ashore and impact a sensitive coastal resource within a day or two unless it is dispersed, there will be a very finite period of time for such considerations suggested in the Proposed Rule.

With respect to the proposed ecological thresholds, although it is constructive to have known benchmarks, it is not clear what decision would be forthcoming if a threshold were exceeded and would certainly be misplaced to derive new thresholds during a response. The documented toxicity of dispersed oil to individual laboratory organisms is by no means a measure of the overall population of that

organism, nor of the ecosystem as a whole, to recover from oil exposure. During a spill, dispersants operations might be stopped because of the threshold of an individual species even though there may be the potential for higher environmental damages beyond that species. During a spill, dispersants operations might be stopped based on a threshold for an individual species . The suggestion of using species sensitivity distributions (SSD) during an active response is a misuse of the method (which requires long periods of time to assess) and is counter to establishing frameworks that are relevant in the dynamic ocean setting.

Rather than requiring the Responsible Party to make this real-time assessment, a CERA/NEBA process (conducted as part of contingency planning) should serve as the basis to make operational decisions on whether dispersants and/or other agents should be used during a response. A CERA/NEBA is the only mechanism that allows a full environmental trade-offs analysis of all response options to be conducted.

XX. Section § 300.913(e) Immediate Reporting Requirements

A. *Reporting requirements should be more flexible.*

The Proposed Rule does not include any indication of what happens if such a requirement to immediately report the information is not met for any reason. From a functional standpoint, the idea of reporting is sound, but the OSC should have the discretion to determine the frequency of reporting.

The Agency is also requesting comments on whether similar requirements should be applied to surface dispersant application. There are already practical and agreed reporting requirements for surface dispersant use and the natural weathering and dispersion that takes place at the surface is fundamentally different than what might occur in the subsurface.

XXI. Section § 300.913(e)(1) Daily Sampling and Data Analyses Reporting

A. *The Agency should not develop requirements for daily authorizations of dispersant quantities.*

This section requires reporting of a deviation of more than 10 percent from the mean hourly volume of dispersant authorized for "*24 hours use*." The Proposed Rule does not require authorization of dispersant use in 24 hour increments. Such restrictive authorization requirements tremendously complicate dispersant operations while providing little environmental benefit. The Agency should not develop requirements for daily authorizations of dispersant quantities, as this decision should be based on the ongoing spill scenario and should be left to the OSC. Dispersant application rates may need to vary to account for dynamic spill conditions, environmental conditions, and other ongoing response operations. Placing daily limits on volumes of dispersants that may be used prevents trained response personnel from utilizing appropriate dispersant application rates. Furthermore, it circumvents the NEBA process that evaluates the best response options that protect the environment as a whole.

We suggest modifying the proposed language as follows:

> *(e) Report to the OSC as soon as practicable or at an interval determined by the OSC:*
> *(1) Deviation of more than 10 percent from the mean hourly dispersant use rate for subsurface application, based on the dispersant volume authorized for 24 hours use, and the reason for the deviation; and*
> *(2) Any threatened or endangered species, or other priority ecological receptors formally identified by the OSC in consultation with the Environmental Unit, that may be exposed, based on dispersed plume trajectory modeling and level of concern information.*

XXII. Section § 300.913(e)(2) Immediate Reporting of Ecological Receptors

A. *Real-time ecological receptor analysis is unrealistic and should be part of a CERA/NEBA process.*

The requirement in this section for real -time ecological receptor analysis is unrealistic. Such efforts need time and broad stakeholder participation to be meaningful. This information should be part of a NEBA process to document the resources at risk of the potentially affected ecosystem. Only through this detailed NEBA process can an adequate understating of overall ecosystem recovery be developed and assessed. Further, the process of overall environmental trade -offs is independent of individual species impacts and should be based at the ecosystem level.

XXIII. Section § 300.913(f) Daily Sampling and Data Analyses Reporting

A. *Daily reporting of water sampling data should only apply to subsea dispersant injection and is not useful for dispersant decision-making.*

This section should be amended to clarify that daily reporting of water sampling data only applies to subsea dispersant injection. Further, the requirement in Section 300.913(f)(3) for a 5-day turnaround time for analysis is unrealistic, given that it could take several days for the sample to be transported to shore and additional time for it to reach a qua lified laboratory and put in the queue for analysis. During DWH, the sheer quantity of samples overwhelmed both the regional and the nation's labs processing abilities. In many cases, the backlog of samples lasted 30 days or longer. A blanket requirement o f five days is unrealistic and should be replaced with a negotiated time that is staggered on the basis of sample data priority for operational decision-making. The data may be useful for NRDA, but not for dispersants decision-making because other monitori ng techniques can better inform decision-making. These water samples will provide information about concentrations in water from 5 or more days previously, but will be irrelevant for operational questions that require near real-time feedback.

We recommend combining this section with the proposed "*Immediate Reporting*" section into one section that notes the importance of reporting in supporting the coordination of response actions. Suggest modifying proposed language as follows:

(e) Reporting

(1) Report to the OSC as soon as practicable or at an interval determined by the OSC any:

(i) Deviation of more than 10 percent from the mean hourly dispersant use rate for subsurface application, based on the dispersant volume authorized for 24 hours use, and the reason for the deviation; and

(ii) Ecological receptors, including any threatened or endangered species that may be exposed based on dispersed plume trajectory modeling and level of concern information.

(2) Report to the OSC at an interval determined by the OSC, water sampling and data analyses collected in § 300.913(b) and include:

(i) Specific hourly dispersant application rate and the total amount of dispersant used for the previous reporting period established by the OSC with concurrence from the EPA representative to the RRT;

(ii) All collected data and analyses of those data within a timeframe necessary to make operational decisions, including documented observations, photographs, video, and any other information related to

dispersant use, based on a reasonable and appropriate timeframe authorized by the OSC;
(iii) Estimates of the transport of dispersed and non-dispersed oil and associated volatile petroleum hydrocarbons, and dispersants, using the best available trajectory modeling.

XXIV. Section § 300.915(a) General Information for any Product Category

A. *The proposed information requirements to be submitted by dispersant manufacturers may reduce the number of dispersant suppliers.*

The proposed requirements for information to be submitted, most likely by dispersant manufacturers, as described in 300.915 (a)(1)-(21), are extensive. API and NOIA are concerned that dispersant manufacturers may not have a sufficient business case to meet all proposed data requirements. The Agency therefore needs to demonstrate that this Proposed Rule will not collapse the market for dispersants and drive away all viable producers from supplying products to meet the need. Further, the Agency should consider developing a total re-listing cost analysis.

XXV. Section § 300.915(a)(5) Safety Data Sheet

A. *A Safety Data Sheet for each component in a product is unnecessary.*

The proposed requirement includes *"A Safety Data Sheet ('SDS') for the product."* We believe this is a reasonable request. However, in the preamble, the Agency solicited comments on the value of requiring an SDS for each individual component. We believe that requiring more than just an SDS for the whole product will add unnecessary burden to the submitter that will not add any protection. The information needed to protect responders will be provided in the SDS for the product as a whole.

XXVI. Section § 300.915(a)(11) Environmental Fate Information

A. *Vendors should not be required to generate new environmental fate information for their products.*

The proposed language in this section should be modified to clearly state that this data should be reported *only* if it is already available. As currently written it is unclear if the vendors are being asked to collect new data for their products to cover these parameters.

XXVII. Section § 300.915(a)(13) Identity and Concentration of Product Components

A. *Vendors should not be required to provide the identity and concentration of product components.*

Some vendors may refuse to submit this detailed data as it may threaten a proprietary advantage. This may result in less favorable/effective product being listed in the Product Schedule. We recommend that reporting should be restricted to naming the ingredients, but should not require that the percentage of each ingredient is disclosed. Although Proposed Rule Section 300.950(b) allows this information to be claimed as confidential business information, the requirement under 300.915(a)(13) still presents challenges because there is no guarantee that the claim of confidentiality will be accepted. This may discourage manufacturers from supplying dispersant products.

XXVIII. Section § 300.915(a)(15) Certification

A. *The requirement that product not contain levels exceeding the National Water Quality Standards is unnecessary.*

The agency is proposing certification, including data, methodology, and supporting documentation, indicating that the product does not contain levels that exceed the National Water Quality Standards lowest aquatic life acute value for the following contaminants: metals reasonably expected to be in the product including arsenic, cadmium, chromium, copper, lead, mercury, nickel, vanadium, zinc; cyanide; chlorinated hydrocarbons; pesticides; polychlorinated biphenyls (PCBs); and polynuclear aromatic hydrocarbons (PAHs).

The National Water Quality Standard Contaminants are defined as concentrations in the water column. Water quality criteria are applicable to receiving waters, not formulated products, and this proposal assumes the appropriate exposure is to full-strength (100%) product rather than a reasonable environmental-realistic exposure concentration.

This requirement is not reasonable because of the enormous dilution that occurs when any product is used in an oil spill situation, and it is not required for other chemicals released into the environment. We believe that the existing requirements to communicate hazardous impurities on product SDSs should be sufficient and suggests removing the requirement.

XXIX. Section § 300.915(a)(19) Annual Product Production Volume

A. *The requirement for vendor production capability is not realistic of the market for dispersants.*

Most dispersants are blends of components and are only manufactured upon request. Capacity is not fixed and varies with available blending tankage, existing business demands, other product orders, and component supplies/shipping constraints. While such data are very useful in a large spill where existing stockpiles could be depleted, the information provided in such a request at the time of the application would not be relevant to a future time when product manufacturing could be required during a response. Further, the development of both U.S. stockpiles and international Dispersant Global Stockpiles were designed to ensure that sufficient, effective dispersants will be available when needed. We believe this requirement should be removed. If not removed altogether, the Agency should consider modifying the language to require dispersant or agent producers to be able to provide such information within 24 hours should the OSC need it and make it a condition of product authorization. If the Agency is concerned about the availability of sufficient volume of dispersants required for operations, it should ensure that the main dispersants products that are used around the world and are extensively studied are included on the National Product Schedule and are available for response in the U.S.

XXX. Section § 300.915(a)(20) Recognition

A. *Design for Environment recognition is unnecessary.*

The Agency asks for recognition from EPA's Design for the Environment (DfE) if applicable. DfE recognition should not be included in the Proposed Rule because it is not required, and the Agency has other ways to advance voluntary programs.

XXXI. Section § 300.915(b)(1) Dispersant Efficacy

A. *The new efficacy test cannot predict field dispersant effectiveness.*

API and NOIA support the Agency's proposed change in efficacy testing from the Swirling Flask Test (SFT) to the Baffled Flask Test (BFT) method. The SFT was a very low energy test and required the use of low viscosity test oil s to achieve significant levels of measured dispersant effectiveness. The BFT is a much more energetic test method thus permitting the use of higher viscosity test oils to achieve higher levels of measured effectiveness with the same dispersant. It is impo rtant to recognize that such laboratory tests can evaluate relative effectiveness between different dispersants, but cannot be used to predict the effectiveness of dispersants in the field on specific oils under specific conditions , such as differences in wave height, possible application in high turbulence subsea or using prop wash in ice - covered waters.

B. *The new test oils may not provide comparable results if new test samples are obtained.*

We also support the Agency's proposed changes in test oils from South Louisiana Crude (SLC) and Prudhoe Bay Crude (PBC) to Alaska North Slope (ANS) and Intermediate Fuel Oil (IFO–120), since these new test oils provide a wider range of properties in the oils to be tested. The agency should note that ANS is a blend of c rude oils produced on the North Slope of Alaska, and the composition and properties have varied through the years as the crude oils produced on the Alaska North Slope have changed . Also, IFO-120 composition is entirely dependent on the crude run at the supplying refinery and their blending needs. However, IFO-120 is a blend of residual fuel components including vacuum resid uum cut with a lighter product to achieve a specified viscosity (i.e. , 120 cSt at 100F in this case), but will vary dramatically in composition between lots at a given refinery and between refineries. It is probable that the varying composition of IFO-120 will result in inconsistent DE results over time. This means that a single batch of IFO-120 will have to be preserved for all future pro duct testing to ensure comparability. The Agency should conduct stability tests of ANS and IFO -120 stocks to ensure they can maintain viable samples over time. The Agency could consider substituting a heavy crude oil rather than this heavy refined product or store a large single batch of IFO -120 with data provided on shelf stability or a provision to account for changes over time.

C. *Testing dispersant efficacy at different temperatures will not provide information that will predict dispersant effectiveness in real world conditions.*

We oppose introducing testing at different temperatures. This new approach implies that laboratory tests represent field conditions in some way which is not the case. The Agency proposes that dispersants should only be approved for the type of oil they worked in the lab and temperatures tested in the lab. This is incorrect as:

- different oils will be treated than the two tested in the laboratory;
- even the same oil will weather with potential effects to dispersant efficacy;
- turbulence in environment will surely be different;
- subsea injection even into the same oil in the Arctic is different because of oil temperature and turbulence level subsea; and
- dispersing oil in ice is also different from lab tests because of the mixing provided by vessel propellers.

In the preamble to this same section, the Agency appears to suggest that the Proposed Rule changes are intended to indicate the likely effectiveness of operational dispersant use on oil spilled in the Arctic or Gulf of Mexico (column 3, 80 Fed. Reg. 3403; both locales are specified). Relatively simple efficacy tests such as the BFT are incapable of simulating the range of parameters and processes that will be encountered and influence real -world dispersant effectiveness. The Rule sho uld clearly state that laboratory efficacy testing is a comparative screening tool for different dispersants under laboratory conditions and should recognize that a ctual dispersant effectiveness on oil treated at sea can vary a great deal from

effectiveness data collected in a laboratory. A dispersant/oil combination that produces an effectiveness result in the BFT laboratory method of 70% might result in >95% dispersion at sea (e.g., in a moderate sea state), and less dispersion in other circumstances (e.g., calmer sea state). Therefore as part of dispersant use authorization, divorced from the product approval process, the availability of suitable mixing energy becomes an operational consideration, coupled with the dilution potential for the dispersed oil. We suggest that reference to specific regions should be deleted from this section.

D. *Dispersant efficacy thresholds listed in the Proposed Rule are too high.*

In the preamble to this section (column 2, 80 Fed. Reg. 3403), the Agency's justification for dispersant efficacy thresholds (DE values) includes:

> "*Three of the eight dispersants tested clearly differentiated themselves from the other five dispersants for having the best DE_{LCL95} efficacy results… This natural break in the results provided the Agency with the basis for the proposed threshold criteria.*"

We believe the proposed thresholds of efficacy for listing on the Schedule are high. The supporting documentation in the docket shows that only six (not eight) dispersants were tested and two of these six dispersants were only tested with one oil. The rationale for selecting the proposed efficacy thresholds for listing on the Schedule is not convincing on the basis of the limited data presented. Data from another EPA study[31] shows significant variability in the results obtained with the BFT. Variability in test results is troubling and this variation could lead to confusing data sets where the same dispersant sometimes "passes" and sometimes "fails" because of the high threshold limit that has been set. While the Agency states it tested the dispersants currently listed and found three of them to be sufficiently effective, the EPA study on the BFT funded by BSEE and published in 2011 suggests otherwise. As shown in Table 3 of this BFT study, Corexit 9500 may not be listed based on published Dispersant Effectiveness (expressed as the LCL95) of 69% and 61% on ANS Fresh and IFO-120, respectively, at room temperature (15° C). This suggests that the recommended thresholds are potentially too restrictive and do not sufficiently take into account the variability of the BFT.

The Agency should reconsider the proposed efficacy thresholds for listing on the Schedule. We suggest that these thresholds could be lowered without any significant impairment to the effectiveness of operational dispersant use at a response. In addition, reporting only the DELCL95 reduces the amount of information available on a product. We recommend that the test average and standard deviation should be provided, which would provide additional information on the precision of the testing.

The Agency makes the following statement in the preamble (column 1, 80 Fed. Reg. 3404):

> "*As a result of this increased mixing energy, better dispersion is realized under conditions more realistic of wave action in the sea*"

We believe that this statement would be more accurate if it were re-phrased as follows:

> "*As a result of this increased mixing energy, higher dispersant effectiveness results are obtained.*"

Since the recent subsea dispersant injection development does not have major implications for the rationale behind the current tests for dispersant product approval, it might be important to recognize that the laboratory test data are sufficient to identify whether a product has appropriate effectiveness,

[31] Venosa and Holder, 2011, Laboratory-Scale Testing of Dispersant Effectiveness of 20 Oils Using the Baffled Flask Test

irrespective of whether the product may be considered for use in surface applica tion or subsea injection (SSDI) scenarios. Additional testing for SSDI approval is not necessary.

XXXII. Section § 300.915(b)(2) Dispersant Toxicity tests and listing criteria

We offer the following comments on the proposed revisions to the toxicity testing and has ordered the comments by specific test.

A. *Standard static acute toxicity testing should be used only for the dispersant alone.*

The Agency proposes that a dispersant shou ld be tested using a modified standard static acute toxicity test for the dispersant with *Americamysis bahia* and *Menidia beryllina* using the 48 hour and 96 hour duration LC_{50} procedure. We agree there is a need for toxicity testing of dispersants to preven t highly toxic (LC50 <1ppm) dispersants from being listed on the Schedule. The proposed toxicity test method, although designed for effluent testing and having no relevance to real -world exposures of marine organisms to dispersants during dispersant use on oil releases, is a convenient way to screen individual dispersants. We agree with this proposal for dispersant-only toxicity tests.

The Agency also specifies a threshold for a dispersant to be listed on the Schedule, *"the LC_{50} being that at the lower 95% confidence interval for all acute toxicity tests must be greater than 10 ppm."* We believe this is a reasonable threshold for the "dispersant only" toxicity test.

B. *Standard static acute toxicity testing of dispersant/oil mixtures does not represent real world exposures.*

The purpose of the proposed toxicity testing of dispersant/oil mixtures is not clear. If the objective of the toxicity testing is to ensure that any dispersant that is to be listed has less than a specified level of toxicity measured in the test, then a dispersant alone toxicity test is sufficient.

We oppose the proposed dispersant/oil acute toxicity testing as a requirement for a dispersant to be listed on the Schedule, for the following reasons:

- It is now widely accepted by scientists that any concerns about the potential for toxic effects on marine orga nisms resulting from the use of modern dispersants should consider the potential effects of dispersed oil, not the dispersant itself. This is logical because dispersant is used at low concentrations, compared to the dispersed oil concentrations produced by their use , and they have much lower toxicity than oil . The potential severity of toxic effects that could be caused to marine organisms by exposure to dispersed oil will be a function of exposure: the concentration of dispersed oil (or oil components) in the water, the duration of that exposure , and oil type and receiving organisms . The Agency proposed dispersant/oil mixture tests will not provide any information about real-world exposure.
- More sophisticated dispersant/oil mixture toxicity testing procedu res, using more relevant exposure regimes, can produce information about the potential for toxic effects on marine organisms by oil dispersed (or oil compounds released in to the water) as the result of dispersant use. Such toxicity studies may be relevant to deciding where and when dispersant use may be authorized, but not to dispersants being listed on the Schedule.
- Toxicity testing of dispersant/oil mixtures (i.e., dispersed oil) usi ng the proposed standard static acute LC_{50} procedures will inevitably discriminate against more effective dispersants which disperse more oil into the water, thereby increasing exposure to aquatic organisms. Such testing does not give any indication of the potential for toxic effects to be caused by dispersant use on spilled oil on the sea surface.

- Previous work on dispersant/oil mixture toxicity testing conducted by the Agency in 2010 (Comparative Toxicity of Louisiana Sweet Crude Oil [LSC] and Chemically Dispersed LSC to Two Gulf of Mexico Aquatic Test Species [published by Hemmer, Barron and Greene]) shows that the proposed 10 ppm threshold (determined with LSC oil, not ANS or IFO -120) with *Americamysis bahia* would not be met by any of the eight dispersa nts tested at that time. Only one dispersant would have met the 10 ppm threshold with *Menidia beryllina*.

In general, the Agency should clearly distinguish between the requirements of (i) the toxicity testing required to assess which dispersants should be listed on the Schedule, and (ii) toxicological studies with appropriate oils, test organisms and exposure conditions that will inform discussions about how the listed dispersants might cause impacts in U.S. waters under the specific circumstances of an oil spill or release. The two toxicity test approaches required are incompatible. Attempting to use the same standard, static acute LC_{50} procedures to provide information or guidance on both aspects will not achieve these goals. Since the proposed dispersant/oil mixture toxicity testing requirements in this section do not address item (ii) above, the Agency needs to clarify the objective and rationale of the proposed acute exposure toxicity testing of dispersant/oil mixtures and should explain how this relate s to the listing of a product on the Schedule.

C. *Testing of dispersants using the developmental assay for the purple sea urchin is not relevant.*

The Agency proposes a requirement for an EPA test protocol involving sea urchin embryos, routinely used in eff luent testing, to be used to assess the potential for a dispersant product to cause adverse effects on the developmental process. The relevance of this test, which was developed for the assessment of wastewater effluents, to dispersant use is highly questi onable. Again, trying to estimate potential real world consequences to dispersed oil exposure should be part of a NEBA decision -based process. The environmental trade-offs of using dispersants in offshore environments to protect sensitive shoreline habitats from oil exposure is being circumvented. Given that the purple sea urchin is found in shallow coastal waters, there is no relevance of applying purple sea urchin data given that dispersant operations are unlikely to occur in these environments. In additi on, the proposed threshold that the LC_{50} must be greater than 10 ppm is not a relevant indicator to inform any potential real -world effects of dispersant use in the environment. Additionally, if the agency applies their proposed toxicity thresholds to the sub-chronic sea urchin toxicity tests, it is likely that many (maybe all) current and approved dispersants in the U .S. will fail the threshold . Some of them could also fail with the sub -chronic *M. beryllina* tests.

We oppose the proposed developmental ass ay and do not believe it should be a requirement for a dispersant to be listed on the Schedule. API and NOIA are also concerned that little experience is available in application of this assay to dispersants, and it is unclear about the intra -lab precision associated with this proposed requirement and the practical challenges that may arise in interpretation of test results.

D. *Sub-chronic 7 -day static toxicity tests for dispersant are not relevant and are unnecessary.*

The Agency states (300.915(b) (2), 80 Fe d. Reg. 3405) that it " *has limited information concerning the possible sub -lethal effects of dispersants currently listed on the Product Schedule* " and that " *This information would also be of value as guidance to regional responders and OSC's on possible ad verse effects on survival and growth of larval fish and invertebrates caused by longer -term exposure to dispersants.*"

Chronic exposure to dispersant alone is not a relevant indicator of risks posed when dispersant is used to mitigate the effects of an oil release. As stated previously, this test was developed and validated for wastewater effluents and is not relevant to the use of dispersants in offshore spill response. Rather than proposing sub-chronic toxicity testing requirements for any dispersant to be listed on the Schedule, the Agency could undertake studies to establish whether this is a valid concern, or whether it could provide useful information under the likely conditions of dispersant use on offshore oil releases.

The Agency's proposal of a No Observed Effect Concentration (NOEC) equal to or greater than 1 ppm listing threshold level for sub-chronic effects is arbitrary. Although this threshold level provides a ten-fold safety factor over the proposed 10 ppm acute toxicity threshold values, it could also be questioned on the same basis; static toxicity tests over 48 to 96 hours do not represent relevant exposure scenarios in the marine ecosystem following offshore applications of dispersant.

We oppose the proposed sub-chronic toxicity testing and does not believe it should be a requirement for a dispersant to be listed on the Schedule. Sub-lethal/sub-chronic studies are not necessary (acute toxicity endpoints are considered relevant) since the global commercial dispersant products are confirmed as biodegradable so that the surfactants and solvent constituents of the dispersant formulation do not persist in the marine environment causing adverse effects. The French dispersant product approval regulator requires a biodegradability test of the dispersant as part of its product approval process. Commercial dispersants commonly stockpiled around the world are therefore readily biodegradable and pass the French test. If a dispersant is approved in France, it must have passed the biodegradation test (AFNOR standard NF T90-346). It is recommended that ready biodegradability of a dispersant be required for approval in the U.S. as well. Therefore any candidate dispersant product would need to be tested for biodegradability, or the dispersant manufacturer could be required to state that the formulation constituents are biodegradable and provide supporting evidence.

E. *Dispersant testing should only be used for screening individual dispersant products.*

If the objective of the toxicity tests are to select less toxic dispersants (based on an aquatic toxicity threshold), then dispersant alone toxicity tests are sufficient and the maximum toxicity threshold of a candidate dispersant should be set if the dispersant is significantly less toxic than the reference oils. The dispersed oil toxicity testing (CEWAF) should not be considered for dispersant use authorization as it discriminates against the effective dispersant products which disperse more oil into the water thereby increasing exposure to the aquatic test organisms. It is also important to note that CEWAF toxicity testing does not simulate actual dispersant use on spilled oil in the marine environment. Additionally, static toxicity tests over 48 to 96 hours do not represent relevant exposure scenarios in the environment following surface application of dispersants.

In summary, the Agency's suggested toxicity testing changes in the Proposed Rule do not take into account the essential role of NEBA in considering the potential benefits and risks of dispersant use as a response option for an offshore oil release. The Product Schedule should focus on the basic testing to compare the efficacy and toxicity of the dispersants against each other with the sole purpose of screening good products that should be listed. The Proposed Rule should not attempt to translate any laboratory data into possible real-world implications. This process should instead be defined by full stakeholder NEBA discussions specific to a given spill scenario.

XXXIII. Section 300.915(c)(1) Surface Washing Agent Efficacy

A. *Surface Washing Agent (SWA) efficacy testing should not be required until the Agency develops its own test protocol.*

In the preamble the Agency has requested comments on available methodologies and whether the rule should identify the specific methodologies to be used until the Agency develops and adopts a new testing protocol for SWA efficacy testing (column 1, 80 Fed. Reg. 3407). The Agency lists two tests by ASTM and Environment Canada as possible interim tests while the Agency develops its own test. There is no evaluation of how the ASTM test compares to the Environment Canada test. As a result, expecting similar results by both tests is unrealistic. The average efficacy in the published data reported using the Environment Canada test may indeed be approximately 30%, but the variation is high and the tests were performed on a Canadian oil using only one test. We believe the Agency should delay requiring SWA efficacy testing until it develops its own test protocol that utilizes a U.S. oil.

XXXIV. Section § 300.915(d) Bioremediation Agent Testing and Listing Requirements

A. *An additional protocol specific to products containing enzymes only would be appropriate.*

The Agency is requesting comments on whether an additional protocol specific to products containing enzymes only would be appropriate. This would consist only of exposure water, weathered oil and the enzymatic product in the concentrations specified by the manufacturer. A new protocol would be useful because effectiveness data from the testing would help determine whether this technology would be beneficial during a response. Many claims may be made about "safer" products to treat oil and thus more credible scientific information is warranted on products approved on the Schedule.

XXXV. Section § 300.915(e) Solidifier Testing and Listing Requirements

A. *Solidifier toxicity testing would be useful.*

The Agency is requesting comments on the need for acute toxicity tests conducted with solidifiers - reference oil mixtures. Toxicity tests with oil may be beneficial as they will help to evaluate efficiency of solidifiers in retaining water soluble hydrocarbons and preventing them from leaching into water. Simple efficiency tests may not provide such data.

XXXVI. Section § 300.915(g) Sorbent Requirements

A. *The proposed Sorbent Product List should not exclude natural sorbent materials.*

The Agency proposes to make available a Sorbent Product List that includes certain publically available non-proprietary sorbent materials. We believe this should be a generic list and should not exclude natural sorbent materials (e.g., hay) that could be obtained locally at a spill site.

XXXVII. Section § 300.950 Submission of Confidential Business Information (CBI)

A. *Full public disclosure of product information may reduce the number of products being made available in the U.S.*

In 300.950 (column 3, 80 Fed. Reg. 3426), the Agency requires everything in the product listing be made public except the actual concentrations of product ingredients. While the Agency may need full disclosure of product formulae, we do not believe the Agency needs to make them public and can accept products with both the ingredients and the concentrations as CBI. Any disclosure of information on product formulae (either ingredients or % concentrations) could put a manufacturer at a competitive disadvantage. This may discourage some manufacturers from listing their product at all and could result in effective dispersant products not being available in the United States.

XXXVIII. Section § 300.955 Addition of a Product to the Schedule

A. *The transition period for currently listed products is too short.*

In 300.955(f), the Agency specifies a 24 -month transition period for currently listed products. This is a very short timeframe given the variety of products listed and depth of technical work (e.g., toxicology studies) required to be contracted, conduc ted, analyzed, and reported. We believe the Agency should extend the transition period to the lesser of five years, the current product expiration date, or after suitable replacement products are listed and commercially available.

We also believe that this section should include a discussion of how currently owned stockpiles purchased in 2010 and 2011 (more than one million gallons of Corexit, in the U .S.) will be tested to comply with the proposed existing stockpile testing requirements of being tested every five years. It is recommended the Agency state that currently held stockpiles will have five years to be re-tested after that dispersant product is listed on the new Product Schedule.

XXXIX. Appendix C to Part 300 Requirements for Product Testing Protocols and Summary Test Data

A. *The dispersant-to-oil ratio (DOR) ratio used in the BFT should be 1:20.*

In Appendix C, Section 2.1 (column 3, 80 Fed. Reg. 3428), the BFT mandates a DOR of 1:25. A DOR of 1:20 has commonly been accepted for use in the U.S. and abroad. Further, the 1:25 DOR may not even be within the manufacturer's recommended use ratio. The Agency needs to ju stify this test requirement. We believe that the DOR for the BFT should be changed to 1:20 to align with the regulatory requirements in 33 CFR 154 and 155 for calculating aerial and vessel dispersant capabilities and to align with standard dispersant spray system calibration. A DOR of 1:20 leads to a 5 gallon per acre dosag e for surface spray systems and is the minimum amount of dispersant to oil slicks that can be visually identified as having sufficient thickness to warrant dispersant application. Having the testing procedure completed for the same DOR would provide more reliance that, when applied, the dispersant will be effective.

B. *The Agency should explain how it will address changes in the refe rence oils as they age and as new samples are acquired for the future.*

In Appendix C, Section 2.3.2, the Agency requires the u se of ANS and IFO -120 as standard reference oils. The Agency is supplying these materials to labs but no data is presented showing shelf stability over time for either and whether stable equivalent materials can be provided in the future. No provision is provided for addressing changes in dispersant efficacy due to changes in composition of ANS (e.g., field production changes, new fields in pipeline blend, etc.).

Also, IFO -120 composition is entirely dependent on the crude run at the supplying refinery an d their blending needs. It is a blend of residual fuel components including vacuum residue cut with a lighter product to achieve a specified viscosity (i.e., 120 cSt at 100 degrees F in this case), but it will vary dramatically in composition between lots at a given refinery and between refineries. If multiple IFO -120 samples were provided from different refineries to a toxicity laboratory, there would be varying results because IFO-120 composition changes from day to day and between refineries. This means that a single batch of IFO -120 will have to be preserved for all future product testing to ensure comparability. The Agency should conduct stability tests of ANS and IFO -120 stocks to ensure they can maintain viable samples over time.

CONCLUSIONS

In conclusion, as evidenced by the extensive substantive comments provided herein, it is clear that much work needs to be done before the Agency should finalize a rule in this key area of spill response, including to make sure that the final rule does not have the unintended consequence of eliminating all or most commercially available and future dispersants for use in spill response. We believe the Proposed Rule will likely result in dispersants being eliminated as a response technique, for the following reasons:

- The Proposed Rule does not recognize that response option decisions must be based on NEBA and that dispersing oil in offshore environments , where it may dilute and biodegrade, is often preferred over the long -term environmental damages that will occur when untreated oil strands and persists in shoreline environments;
- The Proposed Rule focuses on potential risks of dispersants without recognizing the primary purpose of dispersants, which is to rapidly dilute the spilled oil and enhance removal of the o il from the environment through microbial degradation, resulting in reduced VOCs (personnel safety), and a reduction in persistent, stranded oil in shoreline ecosystems (leading to faster ecosystem recovery);
- The Proposed Rule's dispersed oil toxicity tes ting thresholds could eliminate all dispersants that are on the current NCP list and could preclude the listing of future dispersants that are currently in development. Furthermore, the proposed dispersed oil toxicity testing thresholds favor ineffective dispersants over effective products and will mistakenly measure the toxicity of the standard test oils, not of the dispersant product;
- The Proposed Rule adversely constrains the ability of the OSC to make the necessary operational decisions to ensure for an expeditious, efficient, and effective response to mitigate the impacts of a spill to the greatest extent possible;
- The Proposed Rule raises significant issues related to USCG and BSEE facility and vessel oil spill response plan compliance and could resul t in unnecessary and increased environmental impacts and corresponding spill claims as a result of increased damages;
- The Proposed Rule's stockpile re-testing procedures have not taken into account manufacturer recommendations, nor considered scientific publications on the demonstrated, lengthy shelf-life of dispersant products; and
- The Proposed Rule's dispersant monitoring requirements are not focused on collecting data that can provide useful information regarding an ongoing dispersant operation. Instead, these requirements have the potential to delay dispersant use resulting in unnecessary environmental damage.

In addition, the RIA conducted by EPA is substantially flawed. The RIA falls woefully short of EPA's regulatory requirements. EPA fails to provide any quantification of the potential social benefits of the Rule; in fact, the RIA lists benefits associated with avoiding and limiting the impacts of oil spills as potential benefits of this Rule, even though the Proposed Rule has the potential to ag gravate the impacts of an oil spill by limiting response options. As a result, the RIA significantly underestimates the potential costs of the Proposed Rule , while also underestimating the direct costs that the Rule would impose on small businesses, government, and the greater industry.

Accordingly, API and NOIA strongly urge the Agency to consider these comments and suggested revisions to the Proposed Rule to ensure that the best dispersants and dispersant-use practices are available for offshore oil spill response. We recommend that there be further discussions between the Federal agencies and industry scientific and operational experts to add value to further revisions of the Proposed Rule to ensure that the proposed changes to Subpart J align with the latest scientific practices and do not artificially bias against the use of certain response options, such as dispersants.

Furthermore, API and NOIA recommend EPA initiate a Negotiated Rulemaking which has proven to be effective in controversial rulemakings such as this one. At a minimum, we urge the Agency to publish a Supplemental Proposed Rule in the Federal Register for additional public comments before publishing a final rule to ultimately make sure that responders, both from the government and industry, are able to use dispersants in the most effective and expeditious manner, taking into consideration necessary environmental and safety factors, to provide for the best response possible depending on the circumstances of a particular spill event. Given the time it will take to initiate and complete a new rulemaking after the final rule is published it is imperative for the general public and all stakeholders that we get it right the first time.

Thank you for your time and attention to these comments on the Proposed Rule. API, NOIA and our member companies share EPA's goal of establishing sound Dispersants Use Policy and are committed to working with the Agency and the other Federal Agencies involved in oil spill response to ensure the health and safety of people and minimize environmental and community impacts during an oil spill. Please do not hesitate to contact Emily Hague (202-682-8260, Hague@api.org) or Nicolette Nye (202-465-8463, nnye@noia.org) if you have any questions.

Sincerely,

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Analysis of EPA's Regulatory Impact Analysis for the Proposed Revisions to the National Oil and Hazardous Substances Pollution Contingency Plan in 40 CFR 300 Subpart J

Submitted by:

Policy Navigation Group

April 2015

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EXECUTIVE SUMMARY

Policy Navigation Group (PNG) was asked by the American Petroleum Institute to review the Regulatory Impact Analysis (RIA) for U.S. Environmental Protection Agency's (EPA) proposed rule entitled *National Oil and Hazardous Substances Pollution Contingency Plan.*[1] We were asked to evaluate whether the RIA for the proposed rule is consistent with the analytical standards set forward in OMB Circular A-4 and EPA's Guidelines for Preparing Economic Analyses and to perform an independent cost analysis.[2] In our analysis, the proposed RIA has the following principal deficiencies:

- The proposed rule's RIA does not have a consistent baseline that allows the public and policy officials to compare the incremental social costs and the incremental social benefits of EPA's proposed regulation and the individual policy options within the proposal.
- The RIA fails to estimate many policy scenarios and requirements issued in the proposed rulemaking. The RIA thus fails to provide the public a full description of the proposal's impact and undermines the public's opportunity to provide meaningful comment on the proposed rule.
- Further, for the provisions that it does give quantified social cost estimates, the RIA often underestimates compliance costs. The RIA thus underestimates the full social cost to comply and to administer the changes for both private firms and government agencies.
- The RIA does not provide any quantified social benefit estimates either for the proposal as a whole or for its individual options. Moreover, its qualitative description of social benefits is inaccurate and at times actually describes social costs of the proposal. Without estimates of the incremental social benefits of the rulemaking and the proposal's regulatory options, the public cannot provide meaningful comments on the merits of EPA's proposal.
- The RIA's methodology is designed to underestimate the small business impact. EPA's economic impact analysis is inconsistent with EPA's economic analysis guidelines and EPA's requirements under the Regulatory Flexibility Act (RFA).[3] In this analysis, we address some of the RIA's deficiencies and find that the rulemaking could have a significant impact on a substantial number of small businesses subject to the rulemaking.

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[1] 80 FR 3380. Also, U.S. Environmental Protection Agency, *Regulatory Impact Analysis for Proposed Revisions to the National Oil and Hazardous Substances Pollution Contingency Plan Regulations*, December 3, 2014.

[2] U.S. Environmental Protection Agency, *Guidelines for Preparing Economic Analyses*, 2014.

[3] 5 U.S.C. 601-612

- These principal issues are among many aspects of the analysis that are inconsistent with Executive orders on regulatory review, OMB guidance, and EPA's own guidance for economic analysis.[4]

In the RIA, EPA estimates that the total incremental net present value (at 7 percent) social cost of the rule is $7.8 million, or approximately an annualized cost of $0.7 million over 20 years. In our analysis, we find that the estimated social costs are at least three times higher than EPA's estimate. The cost of the major provisions are given below:

Table ES-1. Estimated Social Costs of the Proposal

	NPV at 7% ($)	Annualized Cost over 20 years at 7% ($)
Product Loss Costs	4,000,000-7,300,000	380,000-720,000
Product Testing Costs	5,300,000	500,000
Stranded Capital/Stockpile Value Loss	1,000,000 -- 25,000,000	90,000-2,400,000
Label	180,000	20,000
Recovery Costs	1,700,000	160,000
Monitoring Costs	11,000,000	1,000,000
Preauthorization Costs	200,000	20,000
EPA Costs	800,000	80,000
Totals[5]	24,000,000-52,000,000	2,300,000 --4,900,000

These costs disproportionately burden small businesses. Most producers in this market are small businesses. The RIA did not find a significant effect on a substantial number of these small businesses. However, when we adjust for the more significant costs in Table ES-1 for the provisions the RIA omits and for real cost of capital, 43 percent of small businesses that the RIA analyzed have compliance costs greater than three percent of their 2011 revenues. As shown in Table ES-2, many small businesses face severe impacts. EPA should evaluate whether it must comply with the procedural and other analytic requirements of the Small Business Regulatory Enforcement Fairness Act (SBREFA).

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[4] Executive Order 12866 Regulatory Review, September 30, 1993.

[5] Throughout this report, totals and summaries may be different due to rounding. Reflecting the likely precision of these estimates, figures are presented to two significant digits or less.

Table ES-2. Economic Impact of the Proposal on Small Businesses

Pct of 2011 Revenue	Number of Product Types	Percent of Companies
<3%	17	57%
3%-5%	5	17%
>5%	8	27%

We recommend that EPA correct these deficiencies including undertaking a comprehensive benefit analysis to determine if the rule is in the national interest. Once EPA conducts its analysis of the incremental net social benefits of each proposed provision, it should select the regulatory options that maximize net social benefits. As EPA's economic analysis guidelines states:

> Preparing high -quality economic analysis can greatly enhance the effectiveness of environmental policy decisions by providing policy makers with the ability to systematically assess the consequences of various actions. An economic analysis can describe the implications of policy alternatives not just in terms of economic efficiency, but also in terms of the magnitude and distribution of an array of impacts.[6]

This report discusses the major issues with the RIA in more detail, provides alternative estimates for the proposal's social costs, and recommends how EPA should correct these flaws before reissuing a proposed rule.

[6] U.S. EPA, *Guidelines*, pg. 1-2.

SECTION 1. INCONSISTENCY BETWEEN BASELINE AND POST-REGULATION MARKET SCENARIOS

1.1. BASELINE IS INCOMPLETE AND INCONSISTENT WITH OMB GUIDANCE

The RIA's description of the analytic baseline is incomplete and inconsistent with OMB guidance.[7] There are several problems with the RIA's regulatory baseline:

- The RIA does not establish baseline conditions for many proposed requirements. As discussed in Section 2 below, the RIA does not fully estimate the social benefits and costs of all of the provisions of the proposal. As it corrects the serious deficiency of omitting cost estimates for proposed provisions, EPA must apply a consistent framework for both baseline and post-proposal market conditions in any final rulemaking.

- The RIA assumes a regulatory baseline where all existing manufacturers must pay to test their products. Even in absence of a rule, manufacturers may choose to conduct additional toxicity testing to demonstrate the inherent low hazard of their product and thus to differentiate their products in the market. Oil spill response organizations and responsible parties may also insist on further toxicity testing from producers due to liability concerns. Other jurisdictions also require testing of oil response products. For example, the corresponding European Union (EU) authority, the European Maritime Safety Agency, lists the dispersant testing requirements as of 2014 in the EU member states.[8] Eight EU members require testing of dispersant products. For all these reasons, producers may have some or all of the proposed testing data already available.

Recommendation. EPA should gather additional information on available data not only to prepare a more accurate economic analysis, but also to properly estimate the incremental social costs of this rule. If additional data already exists, EPA could tailor the final rule's requirements to this data and only mandate additional testing where meaningful data gaps exist.

1.2. PROPOSAL DOES NOT HAVE CONSISTENT SCENARIO FOR THE SOCIAL BENEFITS AND SOCIAL COST ESTIMATES

The RIA assumes all existing products comply with the new testing requirements and remain on the market.[9] The RIA also assumes new products are approved by EPA at the same

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[7] U.S. Office of Management and Budget, Circular A-4 Regulatory Analysis, September 2003.

[8] European Maritime Safety Agency, *Inventory of National Policies Regarding the Use of Oil Spill Dispersants in the EU Member States*, 2014

[9] U.S. Environmental Protection Agency, *Regulatory Impact Analysis for Proposed Revisions to the National Oil and Hazardous Substances Pollution Contingency Plan Regulations*, December 3, 2014.

pace as in the past. In other words, society gains the same benefits from products with the same capabilities and efficacy with or without the rule.

If this post-regulatory scenario is true, the rule has few, if any, social benefits.[10] For example, EPA cites reduced product toxicity on marine life as a potential social benefit. However, any potential ecological harm could only possibly be reduced if some existing (and future) products are withdrawn, not used, or disallowed due to the new toxicity testing requirement and due to the new EPA approval process.

However, the RIA's scenario is not the most plausible post-regulation scenario. It is more likely that some existing products will not remain on the approved list after the rulemaking. This outcome is more likely than the RIA scenario for two reasons. First, EPA may decide not to approve some of the existing products. It seems unlikely that EPA would use its scarce resources to promulgate increased toxicity testing standards if it did not have concerns with the toxicity of some existing products.

Second, given the uncertain sales volume, producers may not make a competitive expected rate of return with the additional testing and application expense and the greater risk of disapproval. In this case, they can be expected not to remain on the schedule after the rule is effective.

There is ample evidence that existing producers face a non-zero risk of product disapproval:

- Interest groups and media alleged significant concerns with oil spill product safety during the 2010 SONS.[11,12] Media attention and interest group attention to the issue does not mean that products are toxic per se; however, the threat of media scrutiny dampens producers' interest in bringing products to market. Increased attention is likely to lengthen EPA's approval process and the costs for the producer to respond to media inquiries. For producers, these response costs decrease the rate of return for oil spill products and potentially raise the costs for other company products.[13]

- Many producers are small businesses and need certain rates of return to compensate for their cost of capital and for the opportunity cost to innovate in

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[10] There is the argument that there is an information externality - the producers of the agents have a more concentrated interest and easier access to the potential toxicity information than the purchasers. However, since the producers are mostly small businesses and the purchasers are typically government agencies or large businesses, the purchasers can be expected to have sufficient resources and potential legal liability to overcome any potential information externality.

[11] See for example, http://www.biologicaldiversity.org/programs/public_lands/energy/dirty_energy_development/oil_and_gas/gulf_oil_spill/dispersants.html.

[12] E. Rothsenthal, *In Gulf of Mexico, Chemicals under Scrutiny*, The New York Times, May 5, 2010.

[13] For example, the parent company, and not individual product, may face public criticism.

other markets. Their internal rate of return requirements may preclude further investments once the costs and the risk increase.

In addition to the effect on existing products, the rulemaking is likely to affect the number of future products. By requiring additional testing and approval, the rulemaking increases market barriers. Some new products may not reach the market in the future due to the costs and risks that new products will not pass the new testing requirements.

1.2.1. Social Cost Estimate

To overcome these limitations, we conduct an alternative estimate to illustrate the importance of considering how the rule will raise market barriers. In Section 6 below, we demonstrate that many smaller producers could face a significant economic impact from the rule, i.e., greater than three percent of their annual revenue. As EPA and Small Business Administration guidance suggests, we assume those firms fail or leave the market due to the regulatory costs.[14]

The social cost is the lost value added to society from these products that leave the market. Value added is the remaining revenue after the costs of the inputs (e.g., labor, materials, and energy) are subtracted. Since value added is often proprietary information and varies from firm to firm, we use a recent value added metric for the chemical industry as a whole. From 2013 Bureau of Economic Analysis data, the average valued added in chemical manufacturing is 44 percent of revenue.[15] We assume that each company in the oil spill product market has a value added of 44 percent of revenue.

Once existing firms leave the market, society loses the value added of their products for Subpart J responses. However, in a competitive marketplace, other firms will enter and offer their own products. Since the regulation is not changing the demand for oil spill response products or acting to reduce the likelihood of spills, post-regulatory supply will rise to meet this unchanged demand. Society, therefore, only temporarily loses the availability of some oil response capabilities once existing producers leave the market in response to regulation.

We assume the value added is lost for three years. While three years is an assumption, it is a reasonable one given the proposed rule's timing. Since existing firms have two years to provide EPA with the new test data, if they decide not to reapply, they will likely try to stay on the market for the full two years. Once EPA removes them from the schedule, competitors will then see a market opening. We assume these competitors offer alternatives and submit their applications to EPA one year later.

We recognize other outcomes are possible. Instead of replacing existing products with new products at a 1:1 ratio, fewer new products could enter the marketplace, leaving the

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[14] U.S. Small Business Administration, *Guide for Government Agencies How to Comply with the Regulatory Flexibility Act*, May 2012. See U.S. EPA, *Final Guidance for EPA Rule writers: Regulatory Flexibility Act as amended by the Small Business Regulatory Enforcement Fairness Act*, November 2006.

[15] U.S. Department of Commerce, Bureau of Economic Analysis, *Value Added by Industry*, November 2014

approved products with a greater market share. In this case, compliance costs would be lower than this estimate; however, an additional social cost would be the loss of resiliency to responds to market disruptions. In addition to small firms, large firms may also not submit the new information due to the expense. On an individual product basis, larger firms may not find it profitable to continue.

Recognizing these limitations, based on these assumptions and the revenue information the RIA provides in Exhibit 7-1, Table 1-1 shows our estimate of the social cost of market disruption/transition due to the proposed rule.

Table 1-1. Social Costs from Market Transition

Annual Revenue	Product Type	Assumed Value Added	Social Cost	Discounted Social Cost (7 percent)
$ 62,000	Solidifiers	$ 27,000.00	$ 81,000.00	$ 74,000.00
$ 160,000	Surface Washing Agent	$ 71,000.00	$ 213,000.00	$ 195,000.00
$ 120,000	Solidifiers	$ 53,000.00	$ 159,000.00	$ 146,000.00
$ 200,000	Solidifiers	$ 88,000.00	$ 264,000.00	$ 242,000.00
$ 30,000	Solidifiers	$ 13,000.00	$ 39,000.00	$ 36,000.00
$ 120,000	Surface Washing Agent	$ 53,000.00	$ 159,000.00	$ 146,000.00
$ 150,000	Surface Washing Agent	$ 66,000.00	$ 198,000.00	$ 181,000.00
$ 1,000,000	Dispersant(2) & Surface Washing Agent	$ 441,000.00	$ 1,323,000.00	$ 1,211,000.00
$ 187,000	Surface Washing Agent	$ 82,000.00	$ 246,000.00	$ 225,000.00
$ 290,000	Bioremediation Agent	$ 128,000.00	$ 384,000.00	$ 352,000.00
$ 280,000	Surface Washing Agent	$ 123,000.00	$ 369,000.00	$ 338,000.00
$ 559,000	Bioremediation Agent	$ 246,000.00	$ 738,000.00	$ 676,000.00
$ 120,000	Bioremediation Agent	$ 53,000.00	$ 159,000.00	$ 146,000.00
Total (rounded)				$ 4,000,000.00

Summary	NPV at 7%	Annualized Cost over 20 years at 7%
Product Loss Costs	$4,000,000	$380,000

The social costs for existing products are estimated to be $4 million. If it takes longer for alternative products to enter the market, these costs would be higher. These social costs do not include the loss of efficacy and resiliency to respond to oil spills during this transition period. In Section 2.3 below, we expand this estimate to add the potential product loss social costs from the proposed ban on products containing NP and NPE.

As discussed previously, the regulation will not only discourage existing producers from re-qualifying, but also discourage new entrants by raising the costs and the risks of market

entry. The RIA assumes new entrants arrive after the rule at the same pace as in the baseline, or 10 per year. This assumption is not supported by economic theory. As discussed below in Section 3.4, we predict that application costs would rise by 40-50 percent for surface washing and stabilizer products, 260 percent for bioremediation products, and 120 percent for dispersants compared to current requirements. While we did not find estimates of the supply elasticity in these markets, such significant increases in prices will affect supply. We assume new product applications decrease by 30 percent from 10 to seven per year in response. The cost implications of this change are further discussed in Section 5.

1.2.2. **Recommendation**

Setting the baseline and policy alternative scenario is one of the key foundations of a regulatory analysis. As EPA's Guidelines for Economic Analysis states: "Because the level of detail presented in the baseline specification is an important determinant of the kinds of analysis that can be conducted on proposed regulatory options, careful thought in specifying the baseline is crucial."[16] The RIA has not shown this careful thought. For the final rule, EPA should understand the available toxicity and efficacy data available from producers. EPA should select, at a minimum, alternative market outcomes from the regulation in which some existing products do not return to the approved list. EPA must estimate the incremental social benefits and social costs from more accurate baselines.

[16] U.S. EPA, *Guidelines*, pg. 5-1.

SECTION 2. SOCIAL COSTS NOT INCLUDED IN THE RIA

2.1. STRANDED PRODUCTION CAPITAL AND PRODUCTS

EPA has not estimated the full social costs from a decision not to approve a product or from a producer's decision not to reapply. For the reasons listed in the previous section, it is plausible that some existing products will not be listed in the future due to the new regulatory requirements. In Section 1.2.1, we estimate the temporary lost value to society. The full social costs include the net change in value of the oil spill response product and the stranded human and physical capital used to produce the product. In addition, existing stockpiles of the dis-approved product can be expected to suddenly have lower value due to the disapproval. While on scene coordinators (OSC) could still allow their use during a spill response, the likelihood is lower once the product is no longer approved.

These costs are potentially significant. Stockpiling response products is an essential strategy to minimize oil spills' consequences. Response products must be deployed within hours of an oil spill; there is not time to ramp up production once a spill happens. Stockpiles are substantial and deployed across the globe. For example, a 2014 EU report lists nearly 3,500 metric tons of more than 40 different compounds in EU member state stockpiles.[17]

EPA has recognized this social cost category in other rulemakings. EPA has banned or phased-out products in other environmental programs. For example, EPA regulates the phase-out of ozone-depleting substances (ODS) under the Clean Air Act (CAA). EPA has conducted several major RIAs for the multiple phase-out decisions for ODS use under Title VI of the CAA. For example, in its Section 604 and Section 606 rulemakings, EPA considered the stranded capital and product costs for the ODS subject to the phase-out. EPA also attempted to estimate the social costs of the reduced efficacy of the substitutes.[18]

2.1.1. Social Cost Estimate

We consider the stranded capital and product costs separately. For the stranded capital, there are several considerations in estimating the social cost. First, the regulation does not destroy the labor and physical capital, but blocks its highest valued use. Workers and entrepreneurs can use their knowledge and machinery to make other products and/or the same product. Second, there is a distinct loss as suppliers move their capital from this highest valued use before the regulation to a lower valued use after the regulation. They also have costs as they retrain and retool their equipment for this shift.

This information guides our estimate. At one extreme, if the regulation completely destroyed the economic value of the enterprise, the loss could be estimated as the enterprise values of the firms. Using a standard financial estimation of small business enterprise value, the loss would be three to five times the firm's intrinsic value (i.e., value added). In this

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[17] European Maritime Safety Agency, *Inventory of National Policies Regarding the Use of Oil Spill Dispersants in the EU Member States*, 2014

[18] For a good summary, see Appendix G of *The Benefits and Costs of the Clean Air Act, 1990 to 2010*, U.S. EPA, 1999.

case, the maximum social cost would be $6 million to $8 million. At the other extreme, in a dynamic economy, excess capital and labor is rapidly moved from one use to another, with almost zero transition cost. Since the regulation fundamentally does not change demand, we assume that the stranded human and physical capital costs will be closer to the low end and assume a loss of $1 million.

For the stranded product costs, knowledge of which existing products will not return to the schedule and the amounts of those products are stockpiled is important information. The RIA gives information about the product types that are likely to leave the market due to regulation (see Section 6 below), but not the specific products.

For current amounts in stockpiles, in support of its 2009 vessel response plan (VRP) rulemaking, the Coast Guard complied data on US stockpiles of products listed on Subpart J.[19] Although this data is comprehensive, it is from 1998. The VRP rulemaking required responsible parties to have contracts with oil spill removal organizations (OSRO) or organic capacity to respond to categorical spills.[20] The rulemaking required owners and operators to have response resources capable of applying dispersants. This rulemaking, and subsequent ones, increased the amount of dispersants stockpiled in the United States.

From public information given by OSROs, most advertised dispersant capacity is for Corexit® formulations. While it is very likely that these organizations also stockpile stabilizers, surface washing agents, and potentially bioremediation agents, we did not find a current, comprehensive list of stockpiled substances.

We therefore constructed our estimate under two scenarios: (1) if EPA does not approve Corexit® formulations; and, (2) If EPA does not approve other agents. For the latter, we include these costs in Section 2.4. A potential ban on Corexit® is discussed in this section.

We gathered available data on current and relatively recent reports of Corexit® formulation stockpiles in the United States and globally. Since the chemical is not approved for use in Europe, we found that global stockpiles are located primarily in the United States and Brazil.[21] Table 2-1 gives the organization and the reported stockpile amount.

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[19] U.S. Coast Guard, "Caps" Review of Oil Spill Response Plan Equipment Capability, at https://homeport.uscg.mil/cgibin/st/portal/uscg_docs/MyCG/Editorial/20080725/combined%20document.pdf?id=213ab594d05ab39ffd49d74a2e188eb8246d23ed&user_id=7ee08b44562829c5918c18d08ac87c29.

[20] 74 FR 45004

[21] http://www.oilspillresponse.com/files/OSRL_GDS_Equipment_20Nov2014.pdf

Table 2-1. Estimated Stockpiles of Corexit® Formulations

Organization	Units	gal/unit	Total
MSRC[22]	1	104,000	104,000
Clean Seas Cooperative[23]	1	20,750	20,750
Oil Spill Response[24]	500	264	132,000
Alyeska Pipeline[25]	1	60,000	60,000
Clean Islands[26]	1	37,260	37,260
LOOP[27]	1	33,600	33,600
Clean Gulf Associates[28]	1	34,650	34,650
CISPRI[29]	1	20,570	20,570
Nalco[30]	1	12,000	12,000

Total (rounded)	450,000
Estimated Value	$ 24,000,000

Table 2-1 may overestimate the amount stockpiled since different organizations may have overlapping agreements to use the same stockpile. The estimates may also be underestimates since we could not find publicly available data on the stockpiles of some of the larger OSROs such as NRC.[31] Based on a recently reported retail price from an OSRO of

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[22] https://www-msrc-org-documents.s3.amazonaws.com/major-equipment-list/MSRC_Major_Equipment_List.pdf?AWSAccessKeyId=AKIAIGH7YZWA2HRSZUFA&Expires=1427976589&Signature=lYm5WdnIB7M4XV%2BR1foyl5kUg70%3D

[23] http://www.cleanseas.com/equipment.htm#ABSORBENTS/DISPERSANTS/SPRAY

[24] http://www.oilspillresponse.com/files/OSRL_GDS_Equipment_20Nov2014.pdf

[25] Minerals Management Service, *Assessment of the Use of Dispersants on Oil Spills in California Marine Waters*, S.L. Ross Environmental Research Ltd, June 2002.

[26] http://www.cleanislands.com/

[27] Minerals Management Service, *Assessment of the Use of Dispersants on Oil Spills in California Marine Waters*, S.L. Ross Environmental Research Ltd, June 2002.

[28] http://www.cleangulfassoc.com/equipment.

[29] Minerals Management Service, *Assessment of the Use of Dispersants on Oil Spills in California Marine Waters*, S.L. Ross Environmental Research Ltd, June 2002.

[30] http://www.rrt9.org/external/content/document/2763/904415/1/Nalco%20Dispersant%20Mfgr%20Capability.pdf

[31] http://nrcc.com/our-services/response-services/response-equipment-rental-packages/

$52 per gallon, the reported stockpiles in Table 2-1 are worth $24 million.[32] Since the product is not approved in jurisdictions with sufficient demand or stockpiles to absorb the US stockpile volume, if EPA did not approve its continued use, it is likely that current stockpiles could have little residual value.

In summary, the stranded capital/stockpile value loss is the sum of the loss of enterprise value (assumed to be $1 million) and the potential loss of in value of Corexit® formulations. Since the loss in Corexit® formulations is speculative, we present our estimate for this rulemaking outcome as a range.

Summary	NPV at 7%	Annualized Cost over 20 years at 7%
Stranded Capital/Stockpile Value Loss	$ 1,000,000-$25,000,000	$90,000-$2,400,000

2.1.2. Recommendation

Once EPA establishes a more plausible baseline and post-regulation market conditions, it must account for all social costs that flow from an EPA decision not to approve products currently on the list. These costs include the stranded physical and human production capital, the loss in value of stockpiled products, and any loss in oil spill response efficacy from these decisions. EPA has access to data on oil response product stockpiles that it should use to better estimate the social costs of this rulemaking. EPA's analysis for the final rule should at least be as comprehensive as EPA's approaches in other comparable EPA rulemakings.

2.2. Periodic Revisions of Preauthorization plans

In the proposal, EPA requires Regional Response Teams (RRTs) to review and to update their preauthorization plans for chemical and biological response agents after major oil spills or at least once every five years. The RIA does not estimate the social costs of this proposed requirement.[33]

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[32] http://www.cleancaribbean.org/cart_detail.cfm?x=35

[33] On page 37 of the RIA, under the section entitled "Authorization of Agent Use (part 300.910)," the RIA notes that RRTs and Area Committees are required to provide a preauthorization plan and to follow other requirements. The RIA states: "The incremental burden for members of the RRT and Area Committees requires labor hours to participate in the development of preauthorization plans and to add them to RCPS and ACPs." However, it is not clear whether the extremely low burden the RIA uses (2 hours) is to update the 11 or more preauthorization plans or the title of section, authorization of agent use. In either case, the RIA's estimate is far too low to comply with the proposed requirement.

There is evidence that this provision would have meaningful costs if finalized. Recently, EPA's Inspector General (IG) issued a report finding defects in emergency management planning and contingency planning for oil spills.[34] In response, EPA stated:

> Thus, although we generally agree with the recommendations in this report, and they parallel our efforts, two key obstacles (i.e., limited involvement by other agencies and stakeholders and resources) must be overcome fo r their successful implementation.[35]

The EPA program response to the IG suggests that resources, not interest, is the barrier to updating preauthorization plans. If local governments and other RRT team members are constrained by a lack of resources, then it is not clear how a regulation will alter this reality. The social costs then must reflect this scarcity. The social costs are not limited to the full compensation costs and capital costs necessary to maintain that employee. Unlike private markets, the supply of government officials' time does not increase with greater demand. For this reason, the social costs are the opportunity costs of that official's time - i.e., the social benefits society would gain if the government official spends the time on the next most valuable use of that person's time--enforcement, policy setting, permit approvals, or other public matters. The opportunity cost of this resource - a government agency official's work time - is how EPA's Guidelines for Economic Analysis defines social cost.[36]

2.2.1. **Estimated Social Costs**

Based on the IG's analysis, we estimate the social costs as the resources required to update preauthorization plans more frequently than is current practice. We assume in the baseline that plans are currently revised once every seven years; the proposal requires them every five years or after major oil spills. Based on the IG report finding that some plans had not been updated since the 1990s, a seven year update frequency in the baseline may underestimate social cost. For the post-rule scenario, we use a five year frequency since the other trigger, major spills, are rare and may be expected to lead to preauthorization plan reviews even in the absence of this rulemaking.

The resources required are the time spent revising each plan and the review of each plan by the member organizations. From the agenda of recent RRT meetings, we count the number of distinct Federal and state agencies. In a recent Region 6 RRT meeting, 13 Federal agencies sent staff.[37] We assume that 13 Federal agencies are part of each RRT. States, territories, and Tribes also serve on the RRTs. Among the 11 RRTs (one for each EPA region

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[34] U.S. Environmental Protection Agency, Office of the Inspector General, *EPA Could Improve Contingency Planning for Oil and Hazardous Substance Response*, February 15, 2013.

[35] Ibid, pg. 19.

[36] "Social cost represents the total burden that a regulation will impose on the economy. It is defined as the sum of all opportunity costs incurred as a result of a regulation where an opportunity cost is the value lost to society of any goods and services that will not be produced and consumed as a result of a regulation," *Guidelines*, p.8-1.

[37] http://www.rrt6.org/Uploads/Files/rrt_semi_annual_meeting_summary_2014_November.pdf

and one for Alaska), we assume 56 States, Tribes, and territories send representatives and participants.

We make some assumptions on the effort needed to revise each preauthorization plan. We assume 20 percent of the RRT's member organizations assign one staff person to serve on an RRT subgroup to revise the plan. Each assigned staff person spends 40 total hours in this revision process. Then, each Federal, State, Tribe, and territory agency reviews the draft revised plan in their organization. Three people in each organization are a part of the review to account for the agency's management and likely legal review. Each person reviewing the draft plan spends six hours reading and discussing the revised plan.

For the costs of government staff time, we use median and mean wages for Federal employees and State employees, respectively.[38] To these wages, we apply recent Federal government values for benefits and overhead to the Federal and State hourly wage values.[39] For the reasons discussed above, this estimate of the opportunity cost of this resource likely underestimates its true social value and thus the social cost of this proposed requirement.

We then compare the social costs for the baseline and proposed requirements - the costs of revising the plans on a seven versus five year frequency. We discount the baseline and proposal time stream of costs at a discount rate of seven percent and take the difference between those time streams as the social cost. The net present value of the additional social cost at a discount rate of seven percent is $0.18 million.

Summary	NPV at 7%	Annualized cost over 20 years at 7%
Preauthorization costs	$180,000	$20,000

2.2.2. Recommendation

The RIA must fully include the social costs and social benefits of this provision and calculate the social costs as the opportunity cost of government agency staff time.

2.3. Prohibition on Chemicals and Proposal to Ban EDCs

The RIA does not estimate the social benefits or the social costs for the proposed option to prohibit approving products containing nonylphenol (NP) and nonylphenol ethoxylates (NPE). More significantly, the RIA did not estimate the social benefits and costs of a regulatory option discussed in the preamble, a ban on endocrine disrupting compounds in approved products.

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[38] U.S. Bureau of Labor Statistics, May 2014 State Occupational Employment and Wage Estimates, http://www.bls.gov/oes/current/oes_ms.htm#11-0000.

[39] See for example, U.S. Office of Personnel Management, Federal Investigative Services Division, *White Paper: Support Services Contract Cost Benefit Analysis.*

Banning any compounds will have social costs and potentially social benefits. As discussed elsewhere in these comments, EPA has acknowledged in other rulemakings the loss of income and loss of productive capital from product bans. These costs must be included in the economic analysis of this option.

One notable option EPA lists is to prohibit materials containing endocrine disrupting compounds from the approved list. More broadly, EPA's proposed prohibition - without any stated criteria for the ban - raises significant market uncertainty. Without transparency into EPA's rationale for the prohibition, producers may decide not to pay for testing costs and risk disapproval, reducing the overall efficacy and variety of products available.

However, the social costs are more significant than the loss of net value in the oil spill response market. Bans by a government agency will cast a cloud over the public's perception of the health and safety of a chemical compound. States and companies maintain lists of "toxic" chemicals and have policies that discourage purchasing and use of chemicals on these lists.[40] By banning a substance in oil spill response products, it is more likely that the market for NP and NPE compounds will shrink throughout the overall economy.

Moreover, the incremental social benefits of the proposed NP and NPE ban are uncertain. The RIA did not quantify these benefits. There is reason to believe that any such social benefits will be small. EPA's own test data provides evidence that the proposed bans would have minimal, if any, social benefits. EPA conducted tests of eight dispersant compound-oil mixtures, some of which contained NPE, for their endocrine activity.[41] EPA found:

> None of the eight dispersants tested displayed biologically significant endocrine disrupting activity, with the exception of a weak response for two of the dispersants (Nokomis 3-F4 and ZI- 400) in one of the tests. This estrogenic result is likely not of biological significance. Cell death (degree the dispersant is toxic to living cells) was observed in some tests at concentrations above 10 parts per million. The endocrine and the cytotoxicity screening were conducted at dispersant concentrations from 0.001 parts per million up to 10,000 parts per million. None of the dispersants triggered cell death at the likely concentrations of dispersants expected in the Gulf.[42]

Therefore, although EPA's study had a small sample set, EPA's test data provides no evidence that these oil spill response compounds with NP and NPE pose any inherent hazard that would justify the social costs of a precautionary ban. The preamble does not provide quantified social benefit estimates to support a ban over other policy alternatives.

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[40] See, for example, Maine's "Chemicals of High Concern," at http://www.maine.gov/dep/safechem/highconcern/.

[41] Judson, R. S., Martin, M. T., Reif, D. M., Houck, K. A., Knudsen, T. B., Rotroff, D. M., Dix, D. J. (2010). *Analysis of Eight Oil Spill Dispersants Using Rapid, In Vitro Tests for Endocrine and Other Biological Activity.* Environmental Science & Technology, 44(15), 5979-5985. doi:10.1021/es102150z

[42] http://www.epa.gov/bpspill/dispersants-qanda.html#toxtest8. See also U.S. Environmental Protection Agency, *Toxicity of Louisiana Sweet Crude Oil (LSC) and chemical dispersed LSC to two Gulf of Mexico aquatic test species,* August 2010.

EPA failed to consider and to evaluate policy alternatives to its proposed ban. Under the Executive orders, EPA must evaluate available policy alternatives and select the ones in which the benefits justify the costs.[43] There are several policy alternatives EPA could choose and could evaluate besides a ban on NP and NPE compounds, such as imposing lesser restrictions or requirements. Such options could tailor the social costs more directly to any potential hazard than a product ban. Only through objective, transparent policy analysis of these alternatives can senior officials and the public weigh the options to decide upon the best policy.

2.3.1. **Estimated Social Costs**

We searched PubMED for published, peer-reviewed articles that found evidence of NP and NPE-containing oil spill response agents. The only study that specifically listed these compounds was EPA's 2010 work in response to the SONS.[44] It identified NP and NPE compounds in two listed dispersants, Nokomis 3-F4 and ZI-400. For the purpose of this estimate, we assume these products are subject to the proposal's ban.

Based on our evaluation, the manufacturers of these products are likely small businesses. To estimate the impact of the ban, we use a similar approach as in Section 1.2.1. The social costs are not permanent; the social value of the banned products is eventually replaced by substitutes or through greater sales of other dispersants. In EPA's testing, the most widely-used dispersant did not contain NP and NPE. The social costs, therefore, are limited to the immediate disruption in the market.

Since we do not know the actual sales of these dispersant products, we average the annual revenue from the three small business manufacturers of dispersant products given in the RIA. As done is Section 1.2.1, we use the value added of the chemical industry to estimate the added value from these products. The social cost of the ban is the lost value from these two products for a three year period.

There are several reasons this approach underestimates the likely social cost. First, medium and large companies may have their products banned also. While these companies may have resources to reformulate their products, there still will be social costs to switch to substitutes. Second, our estimate of two banned products comes from our search of the public, peer-reviewed literature. Given the controversy over dispersants during the SONS and EPA's proposed rule, companies have an incentive not to disclose voluntarily NP and NPE ingredients. Third, it is possible that NP and NPE compounds are used in other product types such as surface washing agents. Fourth, the manufacturers of the two dispersants that EPA identified as containing NP and NPE have similar products on the schedule. If these products also contain NP and NPE, they would also face bans under the proposal.

Finally, we do not estimate the social costs of a more sweeping ban on all EDC compounds. EPA's proposal is too vague for the public to understand its implications. First,

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[43] U.S. Office of Management and Budget, Circular A-4 Regulatory Analysis, September 2003, pg. 16.

[44] Judson RS, Martin MT, Reif DM, et al. Analysis of Eight Oil Spill Dispersants Using Rapid, In Vitro Tests for Endocrine and Other Biological Activity. *Environmental science & technology.* 2010; 44(15):5979-5985. doi:10.1021/es102150z.

EPA does not define what it means to be an EDC. Second, EPA does not state whether all compounded named as an EDC are banned as ingredients or whether *de minimis* amounts are permitted.

Since some of these same products may drop out of the market due to the new testing requirement, we structure our NP product ban cost estimate so as not to double count the product loss estimates already presented. In Section 1.2.1, one product is a dispersant. Our range is simply whether this product's loss is included in the $4 million product loss calculated in Section 1.2.1 or is an additional social cost.

Summary	NPV at 7%	Annualized Cost over 20 years at 7%
Product Loss Costs	$4,000,000-$7,300,000	$380,000-$720,000

2.3.2. Recommendation

The RIA must fully estimate the social costs and the social benefits of the proposal to ban NP and NPE compounds in listed products and the alternative to ban EDCs. To comply with the Executive orders and OMB guidance, EPA must also evaluate policy alternatives to banning compounds and select the one with the greatest net social benefits.

2.4. Expired Product Testing and/or Replacement Costs

EPA proposes that stockpiled products must be tested in order to be used after their expiration date. The RIA does not estimate the social costs of this requirement.

This provision has social costs and potentially social benefits. There are two costs: increased product testing costs and potentially the cost of more frequent product shipments to stockpile locations. The social benefits are potentially more effective products due to this requirement.

2.4.1. Estimated Social Costs

Over time organizations that stockpile oil response products will attempt to minimize the cost of this requirement through inventory management. They can change their ordering procedures to better match inventory to use. More significantly, they can shift sales of products past their official expiration date to other markets. Numerous oil spills and cleanups occur that do not involve Federal OSCs. Since many expiration dates are minimal (e.g., "at least two years"), buyers may be willing to accept response products with older shelf lives for a discount or because their spill response need is less. Vendors/organizations that stockpile will, over time, manage their inventories so that products with young shelf lives are reserved for any Subpart J needs and then, rather than retest to verify their continued use, are migrated to other markets.

This strategy can only work if the major stockpiled products have sufficiently long shelf lives. In the Attachment, the recommended shelf lives for each existing product are listed. We assign a shelf life of 20 years to products reporting unlimited shelf life.

Bioremediation products have the shortest average shelf life of less than three years. Dispersants have the longest average shelf life, with the most widely stockpiled dispersant having an unlimited shelf life. While products vary significantly, there are numerous products in each category with relatively long shelf lives to allow inventory management. To the degree there are effects, management of existing bioremediation products could be most vulnerable to the requirement for additional testing.

For these reasons, we model the social cost as a one-time event to shift management of existing inventories. We assume future purchases and sales will direct products with older shelf lives to cleanups outside National Contingency Plan jurisdiction. We also assume that some operators of existing stockpiles and inventories will decide it is more cost-effective to test rather than replace their existing inventory.[45] They may have quantities of the surface washing products with shorter shelf lives than the average nine year life-span of these products. Different stockpiles will have different products; therefore, the testing decision is based on the number of stockpile-product combinations. We assume that operators find 20 stockpile-product combinations where testing is more cost-effective than selling into other markets or discarding and replacing the product. Since dispersants have a much longer shelf live on average, we only assume four stockpile-product decisions to test. Table 2.2 presents the results based on these assumptions. This provision could have social costs of $1.7 million.

Table 2-2 Estimated Costs of Testing Expired Products

Product	Testing Cost	Number of Stockpiles-Products	Total Cost
Bioremediation Agent	$31,800	20	$1,270,000
Dispersant	$13,050	4	$52,200
Surface Washing Agent	$10,300	20	$206,000
Solidifier	$10,300	20	$206,000
Total			**$1,740,000**

In Section 2.1.1, we consider the social cost of managing stockpiles of existing products that are not approved in the post-regulatory scenario. We assume that OSRO and others manage their inventories of surface washing agents, sorbents, and other products to market them to non-NCP spill responses. In Section 5 we combine all of the provisions that require testing into one estimate so that the analysis makes consistent predictions of market dynamics after the rulemaking.

2.4.2. **Recommendation**

EPA must estimate the social benefits and social costs of this provision if it is included in the final rule. EPA should use current data on stockpile amounts and their expiration dates.

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[45] Or they will replace products in response to the rule up to the expected value of testing them to prove their efficacy. We use the testing cost as the upper limit of the cost to replace existing inventories.

2.5. RECOVERY OF AGENTS FROM ENVIRONMENT

In the proposal, EPA adds a regulatory requirement that responsible parties are obligated to recover certain oil spill response products from the environment. When stabilizers, surface washing agents, and sorbents are used, responsible parties must recover them when it is safe to do so. The RIA does not estimate any social benefits or social costs from this provision.

This requirement has social costs and potentially social benefits. For major spills, it is likely the regulation formalizes the likely site-specific orders by the OSC. By making it a regulatory requirement, the major change is that it subjects the responsible party to the full enforcement authorities under the Clean Water Act. The proposal reduces the government's administrative effort to levy fines and penalties.

The marginal increase in the threat of fines and penalties may change responsible party behavior. Responsible parties may put greater effort into recovery operations and respond to reports of agents in the environment after the spill for a longer period of time. More effort at recovery may then yield some environmental benefits on the margin. However, greater human activity - boats, motors, skimmers - in environmentally-sensitive areas may cause damage while recovering response products. The current framework - to take actions which have the greatest net environmental benefits - is more flexible and allows site-specific balancing of ecological, human safety, and response product use.

2.5.1. Estimated Social Costs

As a practical matter, this provision may not have any additional social costs for major Tier I, Tier II, or Tier III oil spills. On scene coordinators effectively order product recovery for these major spills. For larger spills, the greater level of government oversight in both the baseline and the post-regulation may be expected to lead to similar levels of responsible party activity. For smaller spills, the proposed regulatory requirement could increase social costs by diverting resources into product recovery, especially after the immediate spill response. It is difficult to estimate the additional costs in these site-specific decisions.

Since 2000, there have been on average 13 spills per year of between 5,000-10,000 gallons of oil products into U.S. navigable waters.[46] We assume that these spills are large enough that responsible parties use significant amounts of response products, but not so large that they trigger substantial government oversight.

We assume that the regulatory provision triggers an additional $10,000 in response costs per spill of this medium-to-large spill size. Since daily penalties under the Clean Water Act are up to $37,500 per day per violation, a responsible party may be willing to respond to reports of sorbents and other products in the environment after a spill rather than risk a penalty. They may be willing to spend much more than $10,000 per incident to avoid the fines. However, assuming an average per incident additional cost of $10,000, the net present value (at seven percent) social cost of this requirement is $1.7 million.

[46] United States Coast Guard, *Polluting Incidents In and Around U.S. Waters: A Spill/Release Compendium: 1969 – 2011*, December 2012.

2.5.2. **Recommendation**

EPA should articulate in any final rule the specific social benefits and social costs it believes will occur as a result of this provision. Specifically, EPA should explain the value of this requirement as a regulation as opposed to a site-specific/incident-specific decision that maximizes net environmental benefits.

SECTION 3. UNDERESTIMATED COSTS

3.1. MONITORING COSTS

The proposal requires responsible parties to carry out extensive monitoring of dispersant use, oil concentrations, and ecological effects in the event of a major oil spill where dispersants are used. The RIA estimates the cost of these monitoring requirements to be $500,000 per year for an average of one applicable major, off-shore oil spill per year.

The justification for the cost estimate in the RIA is two studies. However, these studies do not replicate the specific proposed requirements and therefore do not provide adequate justification for the RIA's estimate.

3.1.1. Estimated Social Costs

We build a bottom-up estimate of the social costs to monitor and to report the information listed in the proposed rule. We first make some assumptions concerning the duration of the spill. We assume the responsible party conducts active monitoring of the oil plume and dispersant use for 14 days, followed by 21 days of monitoring of the dissolved oil/dispersant mixture in the water column. Modeling of the oil movement in the water is carried out for 14 days; fluorescence monitoring occurs for the entire 35-day period.

Since dispersants are used, the oil spill is assumed to occur off shore, meaning boats are needed to obtain the water samples and to gather data for the modeling. Background samples are assumed to be gathered at four points upwind and downwind of the plume. In-plume measurements are assumed to be gathered at five points, with three different sampling points in the water column at each point. After the 14 day period, monitoring requirements for the next 21 days fall from five locations in the plume to four areas in the affected area. Background samples are still collected.

Field and test blanks are assumed to be analyzed for each background and plume collection point. The proposal also requires the responsible party to submit a quality assurance plan for the sampling and analysis.

Given the need for rapid decisions, the ecological characterization and risk assessment requirements are divided into two phases. For immediate decisions during the 14 day active response, experts gather available information from already-published studies and regional ecological evaluations. This study also sets the acute toxicity level from available data. Once the response moves to the 21-day monitoring of the ultimate dispersion, a more complete, written ecological characterization report is compiled.

The responsible party assigns 10 managerial and technical staff to manage, carry out, and to report compliance with the monitoring requirements during the 35-day event.

Based on this scenario, we gathered publicly-available price data from OSROs and commercial laboratories to build a social cost estimate. The Oil Spill Response Limited

organization has a global price list and cost data for their Florida-based response location.[47] Table 3-1 gives the principal data used in this cost estimate:

Table 3-1. Prices for Oil Spill Dispersant Use Modeling

Fluorescence Tracking[48]	659	GBP/day
Boat Rental[49]	400	GBP/day
Oil Modeling[50]	2,300	GBP/day
Heavy Metal Analysis[51]	350	$/sample
Volatile Organic Compounds and Benzene, Toluene, Ethylbenzene, and Xylenes[52]	250	$/sample
Total Petroleum Hydrocarbons[53]	150	$/sample
Other Parameters[54]	150	$/sample

Based on this scenario, the estimated monitoring costs are approximately $850,000 per spill. This value is substantially greater than the RIA estimate. This estimate does not include insurance costs, contingency costs, surcharges for rapid delivery, sample shipping costs, travel costs, or other logistical needs. Including these costs would increase the social costs of this provision.

Given the frequency of one spill per year, the net present value and annualized value of these social costs are given below.

Summary	NPV at 7%	Annualized Cost over 20 years at 7%
Monitoring Costs	$11,000,000	$1,000,000

[47] http://www.oilspillresponse.com/

[48] http://www.oilspillresponse.com/files/ScaleOfFees2015.pdf.

[49] Ibid

[50] http://www.oilspillresponse.com/files/ScaleOfFees2015.pdf.

[51] See for example, Price List from Water Testing Labs at http://www.wtlmd.com/wastewater-testing-pricing-maryland-md-va-dc-de.php.

[52] Ibid

[53] Ibid

[54] Ibid

3.1.2. Recommendation

EPA should use the available data from OSRO equipment price lists and laboratory sampling costs to build up a more complete cost estimate for all of the required parameters for dispersant modeling in the proposed rule. With a frequency of dispersant use of approximately once per year, there is sufficient existing data and markets to use market prices to estimate the social cost. EPA should also characterize plume size and expected monitoring requirements from past spill responses. Gathering this data will provide more support for a social cost estimate.

3.2. EPA Costs to Administer the Rule

The RIA underestimates the total resource cost for EPA to administer the rule because it fails to include the total compensation and overhead costs to employ Federal agency staff and it omits the time necessary for EPA to administer the new requirements. The RIA's approach does not comply with EPA's obligations under the Paperwork Reduction Act's implementing regulations (PRA).[55]

The RIA use BLS data from 2011 as estimates of the opportunity cost of government employees. There are several problems with this data:

- It is out of date. The BLS has compensation data available through September 2014.[56] Moreover, the General Services schedule of Federal wages for 2014 is also available as it determines EPA employees' paychecks.

- It underestimates total compensation. It is unclear whether the data the RIA uses is for salary and wages only or for total compensation. It appears that the RIA uses a labor cost of approximately $54 per hour for EPA employees.[57] The

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[55] See 5 CFR 1320.3(a)(1):

(1) *Burden* means the total time, effort, or financial resources expended by persons to generate, maintain, retain, or disclose or provide information to or for a Federal agency, including:

(i) Reviewing instructions;

(ii) Developing, acquiring, installing, and utilizing technology and systems for the purpose of collecting, validating, and verifying information;

(iii) Developing, acquiring, installing, and utilizing technology and systems for the purpose of processing and maintaining information;

(iv) Developing, acquiring, installing, and utilizing technology and systems for the purpose of disclosing and providing information;

(v) Adjusting the existing ways to comply with any previously applicable instructions and requirements;

(vi) Training personnel to be able to respond to a collection of information;

(vii) Searching data sources;

(viii) Completing and reviewing the collection of information; and

(ix) Transmitting, or otherwise disclosing the information.

[56] See the Employment Compensation Survey, http://www.bls.gov/ncs/tables.htm.

[57] Calculated from data in RIA, pg. 36.

RIA should use total compensation measures to account for Federal employee benefits such as pension, health care, disability insurance, and personal leave. This information is readily available in EPA budget documents, in OMB budget memoranda, and from the Office of Personnel and Management.

- The total resource costs for government services is more than the total compensation cost of labor. Government workers do not work in the open air. They need buildings, electricity, computers, telephones, water, and many other overhead costs. Estimating the overhead costs for government employment is a routine practice in Federal budgeting, in program evaluation, and for compliance with the Paperwork Reduction Act.[58]

Further, the RIA does not include the costs for EPA to administer certain provisions of the rule. In addition to the costs to participate in the RRT and AC meetings included in Section 2.2.1, the RIA fails to include the EPA resources necessary to remove a product from the listing schedule.

3.2.1. Estimated Social Costs

We estimate the social costs of EPA resources to oversee the proposed requirements. Although it is likely an underestimate, we retain the RIA's estimate of 26.5 hours for EPA staff to review each product application. Interestingly, the RIA estimates producers of existing products spend only 7.7 to 27.7 hours to submit the required testing data to EPA. Thus, by the RIA, EPA's review time is longer - up to three times longer - than the time needed to prepare the information. If this ratio is correct, EPA's review process could take over 100 hours. We also agree with the RIA's estimate that the same number of hours is needed when a manufacturer submits new testing data after a product has been reformulated.

However, the RIA assumes EPA spends only 2.5 hours reviewing resubmissions of data from applicants. Since EPA has only 90 days to review an application, it is likely to extend its review by asking applicants for additional information. It seems implausible that EPA would need less than 10 percent of its estimated total review time to review resubmitted data. We use an estimate of 20 hours. We also double the amount of time the RIA estimates to review an appeal of a product disapproval. We believe other divisions of EPA, including the Office of General Counsel, will support the Administrator's review of an appeal and a final Agency decision.

For the opportunity cost of EPA personnel time, we use the fully-loaded employment costs. In FY13, the median EPA employee salary was nearly $108,000, or nearly $55 per hour.[59] We apply OPM's benefit and overhead percentages to calculate a fully-loaded rate of

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[58] 5 CFR 1320.3(a)(1).

[59] U.S. Office of Personnel Management, *Salary Information for the Executive Branch Fiscal Year 2013*, April 2014.

$81.50 per hour.[60] This estimate is a slight underestimate of the likely median wage due to general schedule increases since FY13.

Using the post-regulation rate of existing product submissions and new product submissions, we calculate a net present value (at seven percent) of $0.81 million in social costs for EPA to administer the testing requirements.

3.2.2. Recommendation

The RIA should include costs of EPA to carry out all the proposal in its entirety. EPA should include the total compensation cost for government personnel and the overhead costs necessary to house and to equip these employees so that they can perform their new tasks under this proposed regulation. EPA must include these costs and the resources to oversee all of the provisions of the rule in subsequent analyses.

3.3. Testing and Product Listing Costs

The RIA underestimates the costs an organization expends to understand and to comply with this regulation. The RIA assumes essentially that one or two individuals are solitary actors that read, plan, and implement a regulation for a company. In reality, many employees and managers in an organization will be involved in a product's application.

In addition to underestimating the costs for producers, the RIA underestimates the costs to understand, to comply, and to submit product applications. For example, the RIA does not include these costs:

- Laboratory Personnel Understanding the Rule. The RIA does not include the cost for laboratory personnel either within the manufacturer or at a commercial lab to understand the new testing requirements.
- Legal Personnel Time. It seems implausible that a company would file an important submission to EPA without it undergoing legal review. The submission could determine the entire market for a product.
- Response Costs. Often EPA staff will seek clarifications from respondents on their data submissions to the agency. Seeking clarifications in lieu of outright denials/approvals conserves resources for both EPA and respondents. Oddly, the RIA includes EPA's cost to review these re-submissions, but not those of the respondents to prepare them.[61]

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[60] See for example, U.S. Office of Personnel Management, Federal Investigative Services Division, *White Paper: Support Services Contract Cost Benefit Analysis.*

[61] Even if the RIA estimates the full resource cost for the respondent to submit a "perfect" application that does not require clarification, the respondent is not in control of EPA staff comprehension of the submission and their ability to ask questions, triggering respondent costs.

3.3.1. Estimated Social Costs

In Table 3-2 below, we present alternative labor estimates for producers to comply with the rule that addresses the RIA's limitations. We include hours for all the provisions required of producers, including ones the RIA omits like the product label redesign.

Table 3-2. Estimated Skilled Labor Resources Producers Require to Comply with Proposal's Testing Requirements

(Hours)

	Labor Hours Needed		
	Managerial	Technical	Clerical
Hourly Compensation	$ 55.56	$ 55.56	$23.57
Understand Rule	4	8	2
Contract for Tests	2	16	4
Manage Tests	0	8	8
Prepare Documentation	4	40	16
Design Label	0.5	8	8
Legal Review	4	0	2
Revise Documentation	2	4	45
Respond/Follow up	4	16	8
Total (Hours)	20.5	100	93
Total ($) including overhead rate of 17 percent			$10,400

We include hours for essential tasks to prepare the required information. First, the producer must find and contract with an independent certified laboratory for the efficacy and toxicity testing. They must manage the outside laboratory's performance and review the results. In addition, the producer must prepare the written submission to EPA and follow up with any EPA questions.

For labor rates, we use the BLS Employer Cost for Employment Survey data for December 2014.[62] We use the same overhead rate as the RIA, 17 percent, and use the RIA's relative proportion of the application costs among dispersants, solidifiers, surface washing agents, and bioremediation agents. We use these labor costs for product testing in the multiple provisions in the proposal that require product testing.

[62] http://www.bls.gov/ncs/ect/sp/ececqrtn.pdf.

3.3.2. Recommendation

EPA should include all the costs to understand, to explain, and to incorporate the proposed rule into their organizations' operations. EPA is required to perform this analysis under the PRA regulations. In addition, from past applications, EPA should review whether, and if so, the scope of additional information requests it gave to applicants and include these costs in the any final rule. These costs must be assigned to the regulation.

3.4. Limitation To Tested Waters and Efficacy and Toxicity Testing Costs

In the proposal, EPA limits use of a product to the water characteristics of its toxicity and efficacy testing. Product data with just freshwater test data are limited to freshwater uses, for example. The proposed rule expands the number of efficacy tests for bioremediation agents and alters the efficacy test for dispersants. The proposed rule adds new toxicity tests for dispersants and freshwater/saltwater tests for bioremediation, solidifiers, surface washing agents, and other products.

The RIA makes a global assumption that producers chose just one testing environment, salt water or freshwater, and bear the costs for only one test protocol. Rather than chase market demand and maximize net profits, producers instead make their marketing decisions based on the testing cost. The RIA's assumption violates basic economic theory and thus underestimates social costs.

EPA's own data demonstrates this point. Producers list which of the current required toxicity and efficacy testing they have conducted. This information is available on the Subpart J schedule. Except for the bioremediation products, all other products list test data in both the reference saltwater and freshwater species. For bioremediation products, 27 products do not list any information; 21 report test data in both species. In other words, producers currently want to sell in both markets due to the demand for oil spill remediation in both waters. The rulemaking does not change this market demand.

3.4.1. Estimated Social Costs

Since producers currently market to oil spill response demands in both freshwater and saltwater environments, we assume existing and new product manufacturers will pay for the new efficacy and toxicity tests required for both environments. Using the RIA's testing costs, the total laboratory testing cost per product is listed in Table 3-3.

Table 3-3. Testing Costs by Product Type

Dispersants	$13,050
Bioremediation	$31,800
Solidifiers,	$10,300
Surface Washing	$10,300

We use these testing costs to estimate the application cost and the economic impact of the proposed rule. The labor and laboratory testing costs are combined and presented in Section 6.

3.4.2. Recommendation

Consumer demand will drive producers' decisions on marketing, not the testing requirements of the rule. EPA should align its economic analysis to observed producer decisions and thus expect increased testing costs rather than the estimates found in the RIA.

3.5. Retesting Whenever Chemical Composition Changes

In the proposal, producers would be required to retest its products and submit the results to EPA whenever they change the chemical compounds or their concentrations in an approved product. The proposal calls for comment on an option for allowing *de minimis* changes in concentrations or formulations, but proposes retesting for any change. The RIA assigns the same amount of cost to this requirement as it does for a new product application. For the reasons discussed in Section 3.3 above, the RIA underestimates social costs.

3.5.1. Estimated Social Costs

For a cost estimate, we assume that on average one product each year undergoes a change that triggers the requirement to retest. With more than 100 products and product lines on the Subpart J list, the RIA's assumption seems reasonable that approximately one product would undergo a change each year to trigger retesting.

Since we do not know which product type will undergo change, we take an average of the estimated testing costs for the four product types. Therefore, if one product triggers the retesting requirement per year, the net present value (at seven percent) of this proposed provision is $1.7 million/year.

3.5.2. Recommendation

EPA should articulate in any final rule the specific social benefits and social costs it believes will occur as a result of this provision. Specifically, EPA should explain the value of this requirement as opposed to a performance standard that makes manufacturers responsible to demonstrate that formulation changes do not affect toxicity or efficacy.

3.6. Product Label Redesign

In the proposed rule, applicants must submit a sample product label that includes the manufacture and expiration date, the storage conditions, and the mandatory disclaimer language. Producers must not only change the description on the label on the physical product, but also the description on any promotional material, websites, and other presentation forms. The RIA includes the social cost of submitting the sample label in its overall producer cost estimates.

EPA has estimated the social costs of mandatory labeling and labeling changes in multiple rulemakings.[63] The social costs include the one-time labor and capital costs to craft the new label design, any on-going incremental costs to produce the new label, and costs to digitize the information for electronic media.

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[63] See, for example, U.S. Environmental Protection Agency, *Final Technical Support Document, Fuel Economy Labeling of Motor Vehicles: Revisions to Improve Calculation of Fuel Economy Estimates*, EPA420-R-06-017, December 2006.

3.6.1. Estimated Social Costs

We include additional labor hours to account for the changes to a producer's website and marketing materials. Table 3-4 shows the labor costs for this requirement. We add $500 for each producer in capital and material costs to create the new label.

3.6.2. Recommendation

In any final rulemaking RIA, EPA should draw upon other EPA analyses to estimate the labor hours and capital cost requirements to redesign labels and marketing materials.

3.7. Accredited Laboratory Requirement

The proposed rule requires test data be generated from an accredited laboratory. In the baseline, companies could use any laboratory or generate the information in their own laboratories. While the RIA states that it gathered costs from laboratories to conduct the tests, it is unclear whether this cost information is from accredited labs.

3.7.1. Estimated Social Costs

If the RIA data is from accredited labs, then the costs of this requirement is included in the test costs used in the RIA. If it is not, then the RIA's costs for each test is likely underestimated.

3.7.2. Recommendation

In any final rule, the accompanying RIA should disclose the sources of its information so that policy officials and the public can evaluate the incremental social benefits and social costs of this provision as required by the Executive orders on regulatory review.

SECTION 4. THE RIA'S SOCIAL BENEFIT DESCRIPTION IS INACCURATE AND DOES NOT COMPLY WITH EPA'S REGULATORY ANALYSIS REQUIREMENTS

Fundamentally, the RIA fails to comply with the Executive orders on regulatory review because it does not quantify the potential social benefits either in dollars or in any other metric.[64]

Even more troubling, the RIA lists social benefits that are irrelevant to this rule and, if applicable at all, are actually social costs. The RIA lists qualitative benefits from avoiding oil spills such as lost tourism revenue and enjoyment. However, this rulemaking has the potential for aggravating the potential damages from oil spills. To the degree this rulemaking reduces chemical and biological agent availability and innovation, the nation will have less tools available to fight oil spill damage. A possible, but hopefully remote, outcome of this rule is that future oil spill damages would be more severe than they would be if the rule did not occur. Therefore, the potential social costs from oil spills should be considered, if at all, as a social cost of this rulemaking, not a benefit.

The RIA also does not quantify the marginal social benefits from each provision and each regulatory option. Under Circular A-4, agencies are required to estimate the marginal social costs and benefits for each regulatory provision.[65] Since this rulemaking proposes multiple changes to Subpart J, the public and policy officials cannot determine which provisions have net social benefits and which provision provides the most social benefits.

As discussed in Section 1, the other fundamental problem is that the RIA does not have a consistent baseline and post-regulatory alternative to allow a consistent measurement of social benefits. We present our recommended post-regulatory scenario in Section 5.

Recommendation. The RIA does not comply with the basic requirements of a social benefit estimate. First, it must define clearly what the social benefits are from this rulemaking and describe the real social goods that increase from the baseline to the post-regulation scenario. Second, an RIA for any final rulemaking must attempt to quantify the incremental amount of these social goods that can be attributed to this rulemaking and to the specific provisions of the rulemaking that will generate these social goods. Finally, EPA must use the established literature to value the gains, if any, in ecological services from the rulemaking.

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[64] OMB, Circular A-4, pg. 27.

[65] OMB, Circular A-4, pg. 7.

SECTION 5. TOTAL SOCIAL COSTS

5.1. POST-MARKET CONDITIONS

As discussed in Section 1, EPA's economic analysis guidelines emphasize the need for a clear and consistent market baseline and description of post-regulation market conditions. This guidance is especially relevant for this rulemaking because of the interaction of the requirements, the effect on regulated parties' behavior, and the ultimate social cost of the rulemaking.

This section describes a potential sequence of actions flowing from EPA's promulgation of the final rule. The social cost of these actions, and the time they occur, is the structure of our social cost estimates:

- In year 1, EPA promulgates the final rule.
- In year 1 and 2, existing products remain on the market. At the end of year 2, 13 existing products do not submit testing data due to high costs. The value of 13 comes from the results of the economic impact analysis in Section 6. Therefore, the number of existing submitted application is 13 less than the baseline. It is likely that these 13 products include the products banned due to the NP and NPE formulation ban.[66]
- Starting in year 1 and continuing, the rate of new product applications falls from 10 per year to 7 due to increased costs and market barriers
- In year 3, 13 new product applications are submitted to fill the baseline market demand.
- In years 1-3, 64 expired product applications come as suppliers adjust their inventory. We assume 24 in year 1 and 20 each in years 2 and 3.
- One new application appears each year for product reformulation.

[66] In other words, the social costs of the product loss from the banned products are estimated. However, we assume that for the one or two potential products subject to the ban, only one new product is added in the future.

Table 5-1. Pace of Product Applications

	Year						
	1	2	3	4	5	6	7
Baseline							
Existing Products	109						
Annual New Products	10	10	10	10	10	10	10
New Estimate							
Existing Products	59	27					
Expired Products	24	20	20				
Annual New Products/New Formulations	7	8	8	21	8	8	8
Total	**90**	**55**	**28**	**21**	**8**	**8**	**8**

5.2. ADDITIONAL SOCIAL COSTS

5.2.1. Additional Social Costs from this Analysis

Based on the information in Table 5-1, we can integrate the social cost estimates for the different provisions of the rule. Table 5-2 gives the net present value and annualized social costs for the provisions with estimates in this analysis. As in Table 5-1, the social cost for testing includes the costs for expired products, for new products, for retesting of existing products, and for impact of existing products dropping out of the market.

Table 5-2. Additional Social Costs of the Proposal

	NPV at 7% ($)	Annualized Cost over 20 years at 7% ($)
Product Loss Costs	4,000,000-7,300,000	380,000-720,000
Product Testing Costs	5,300,000[67]	500,000
Stranded Capital/Stockpile Value Loss	1,000,000 -- 25,000,000	90,000-2,400,000
Label	180,000	20,000

[67] This number is the net present value of the weighted annual testing cost multiplied by the number of products per year that is given in Table 5-1. The weighted annual testing cost is that year's mix of dispersant, solidifiers, surface washing, and bioremediation products that submit testing information. For existing products, the each yearly mix of different product types is proportional to the number of each type on Subpart J and the number of each type subtracted/added to the additional testing cost. As discussed in Table 2-2, different product types also have a different frequency of expired product testing. This proportion thus changes the number of each product type tested per year and thus the weighted annual testing cost.

Recovery Costs	1,700,000	160,000
Monitoring Costs	11,000,000	1,000,000
Preauthorization Costs	200,000	20,000
EPA Costs	800,000	80,000

Totals[68]	24,000,000-52,000,000	2,300,000 --4,900,000

5.2.2. **Other Costs from the RIA**

The RIA included social costs estimates for other proposed requirements that are not re-estimated in this analysis. The RIA estimates the costs for some sorbent manufacturers to submit technical data to be considered on a Sorbent Product List. The RIA estimates the annualized cost to be approximately $15,000. Adding this provision to the costs in Table 5-2 does not materially change the social cost estimate.

[68] Throughout this report, totals and summaries may be different due to rounding. Reflecting the likely precision of these estimates, figures are presented to two significant digits or less.

SECTION 6. THE RIA UNDERESTIMATES THE SMALL BUSINESS IMPACT

The RIA's methodology underestimates the rule's economic impact on small businesses. The proposed rulemaking may have a substantial effect on a significant number of small businesses in the oil spill response product market. EPA should evaluate whether it must comply the procedural and other analytic requirements of the Small Business Regulatory Enforcement Fairness Act (SBREFA).

EPA's methodology underestimates the economic impact in two principal ways. First, as discussed above, it underestimates the direct labor and capital costs to comply with the rule. Second, it underestimates the cost of capital in at least two ways: (1) by assuming owners can finance testing requirements over a longer time period than is likely; and, (2) by underestimating the cost of capital for small businesses. Each of these issues is discussed below.

First, firms producing existing listed products only have two years to reapply and to resubmit the newly-required information. Yet, the RIA annualizes a firm's costs over a 20 year period. It is highly unlikely that a small business could borrow funds over a 20 year period for a product that may only have a limited market for two years. Nor would rational business owners with high opportunity costs for capital tie up their capital for such a long time period. A more realistic financing period is two years.

Second, as EPA's economic guidelines state for economic impact analyses, EPA should use the private cost of capital for small businesses.[69] EPA uses the social cost of capital established by OMB. However, due to the increased loss risk, capital lending institutions or capital investors charge higher effective interest rates for loans to small businesses. The Federal Reserve conducts routine survey of small business lending conditions.[70] In recent months, they have found completed, Small Business Administration-backed small business loans at an interest rate of between 5.1 and 5.6 percent. However, the average effective borrowing rate across all small businesses is much higher. As firms report, if they do not qualify for commercial credit, they use instruments with higher interest rates such as credit cards, trade credit, or debt consolidation. The average effective borrowing rate across all small businesses and all credit needs is substantially greater than seven percent.

We present an alternative estimate of economic impact to small business as an illustration of how large the impacts are likely to be. Table 6-1 shows the testing and the labor costs for existing producers to comply with the proposed rule using different discount rates and annualization periods.

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[69] EPA, *Guidelines*, pg. 9-16.

[70] Federal Reserve Banks of New York, Atlanta, Cleveland and Philadelphia, *Joint Small Business Credit Survey Report, 2014*, February 2015.

Table 6-1. Annualized Compliance Costs for New Testing Requirements, RIA Estimate and Revised Estimate

	Testing Costs	Estimated Labor Costs	RIA's Total Costs	Estimated Total Costs	Annualized at 7% over 20 Years	Annualized at 7% over 5 Years	Annualized at 7% over 2 Years
Dispersants	$13,050	$10,398	$14,240	$23,448	$1,344	$3,473	$12,969
Bioremediation	$31,800	$7,728	$14,042	$39,528	$1,325	$3,425	$21,863
Solidifiers,	$10,300	$6,412	$5,986	$16,712	$565	$1,460	$9,243
Surface Washing	$10,300	$5,077	$5,880	$15,377	$555	$1,434	$8,505

Table 6-2 replicates Exhibit 7-1 of the RIA to present the testing and application costs as a percentage of the firm's annual sales. For economic impact analyses, if a regulation's cost to a firm is three percent of its revenue or more, the general conclusion is that the firm is at risk of closure.[71] The RIA reviewed 30 of the 35 small businesses and found that none exceeded the three percent significant impact threshold. However, using a more realistic financing period and more realistic estimates of costs, 43 percent (13/30) of the firms would face compliance costs greater than three percent of their annual 2011 sales. We recognize that Table 3 uses 2014 values for labor costs and 2011 data for revenue. In any final rule, EPA should update the revenue estimates and recalculate the economic impact using costs and financing options that reflect market conditions.

[71] U.S. Small Business Administration, *Guide for Government Agencies How to Comply with the Regulatory Flexibility Act*, May 2012. See U.S. EPA, *Final Guidance for EPA Rulewriters: Regulatory Flexibility Act as amended by the Small Business Regulatory Enforcement Fairness Act*, November 2006.

Table 6-2. Economic Impact on Small Businesses

Company	Number of Employees	Annual Revenue	Product Type	Annualized Cost	Pct of Revenue
1	1	$ 62,000	Solidifiers	$9,243	14.9%
2	1	$ 1,000,000	Surface Washing Agent	$8,505	0.9%
3	1	$ 160,000	Surface Washing Agent	$8,505	5.3%
4	2	$ 2,500,000	Bioremediation Agent	$21,863	0.9%
5	2	$ 120,000	Solidifiers	$9,243	7.7%
6	2	$ 200,000	Solidifiers	$9,243	4.6%
7	2	$ 30,000	Solidifiers	$9,243	30.8%
8	2	$ 10,000,000	Solidifiers	$9,243	0.1%
9	2	$ 120,000	Surface Washing Agent	$8,505	7.1%
10	3	$ 280,000	Surface Washing Agent	$8,505	3.0%
11	3	$ 1,300,000	Surface Washing Agent	$8,505	0.7%
12	3	$ 150,000	Surface Washing Agent	$8,505	5.7%
13	3	$ 5,000,000	Surface Washing Agent	$8,505	0.2%
14	4	$ 559,000	Bioremediation Agent	$21,863	3.9%
15	4	$ 1,000,000	Dispersant(2) & Surface Washing Agent	$34,442.26	3.4%
16	4	$ 730,000	Solidifiers	$9,243	1.3%
17	4	$ 500,000	Surface Washing Agent	$8,505	1.7%
18	4	$ 187,000	Surface Washing Agent	$8,505	4.5%
19	4	$ 350,000	Surface Washing Agent	$8,505	2.4%
20	5	$ 5,000,000	Bioremediation Agent	$21,863	0.4%
21	5	$ 290,000	Bioremediation Agent	$21,863	7.5%
22	6	$ 120,000	Bioremediation Agent	$21,863	18.2%
23	6	$ 15,000,000	Dispersant(1) & Surface Washing Agent	$21,473	0.1%
24	6	$ 500,000	Surface Washing Agent	$8,505	1.7%
25	6	$ 5,000,000	Surface Washing Agent	$8,505	0.2%
26	7	$ 660,000	Surface Washing Agent	$8,505	1.3%
27	8	$ 3,054,214	Bioremediation Agent	$21,863	0.7%
28	8	$ 2,500,000	Surface Washing Agent	$8,505	0.3%
29	8	$ 1,500,000	Surface Washing Agent	$8,505	0.6%
30	9	$ 2,500,000	Surface Washing Agent	$8,505	0.3%

ATTACHMENT: SHELF LIFE OF LISTED AGENTS[72]

Product	Type	Shelf Life	Assumed Shelf Life (years)
AGROREMED (see SPILLREMED (MARINE)®)	Bioremediation Agent	one year	1
BILGEREMED (see SPILLREMED (MARINE)®)	Bioremediation Agent	one year	1
HYDROREMED (see SPILLREMED (MARINE)®)	Bioremediation Agent	one year	1
INIPOL EAP 22	Bioremediation Agent	one year	1
MUNOX SR®	Bioremediation Agent	over one year when stored at temperatures from 35°F to 95°F	1
OILCLEAN w/ACTIVATOR (see PRO-ACT)	Bioremediation Agent	one year	1
PRISTINE SEA II	Bioremediation Agent	As a dry bacterial blend, the shelf life is 1 year, and as a liquid bacterial mixture, the shelf life is 6 months	1
PRO-ACT	Bioremediation Agent	One year	1
SPILLREMED (INDUSTRIAL)	Bioremediation Agent	one year	1
VAPORREMED (see SPILLREMED (MARINE)®)	Bioremediation Agent	one year	1
BILGE CLEAR (see S-200)	Bioremediation Agent	1½ years if kept in the unopened original container	1.5
NANOBITE (see S-200)	Bioremediation Agent	1½ years	1.5
OIL GONE EASY (see S-200)	Bioremediation Agent	1½ years if kept in the unopened original container	1.5
S-200	Bioremediation Agent	1½ years	1.5
S-200C (see S-200)	Bioremediation Agent	1½ years	1.5

[72] Information compiled from individual products listed on schedule as of March, 2015. http://www2.epa.gov/emergency-response/alphabetical-list-ncp-product-schedule-products-available-use-during-oil-spill.

S-200 OIL CLEAN (see S-200)	Bioremediation Agent	1½ years	1.5
S-200 OIL GONE (see S-200)	Bioremediation Agent	1½ years	1.5
SHEENCLEAN (see S-200)	Bioremediation Agent	1½ years	1.5
Z-11 (see S-200)	Bioremediation Agent	1½ years	1.5
BIO-REGEN HYDROCARBON (see SOIL RX)	Bioremediation Agent	Shelf life exceeds 2 years when properly stored in the unopened original container	2
BIOREM-2000 OIL DIGESTER™	Bioremediation Agent	Two years when stored within the storage temperature range in the original container	2
BIOREM-2000 SC (see BIOREM-2000 OIL DIGESTER™)	Bioremediation Agent	Two years when stored within the storage temperature range in the original container	2
JE1058BS	Bioremediation Agent	2 years if stored in a cool dry area	2
REMEDIADE™	Bioremediation Agent	unopened is 2 years	2
SOIL RX	Bioremediation Agent	two years	2
SP 7010 (see REMEDIADE™)	Bioremediation Agent	two years	2
SUMP SAFE BIO-RECLAIM	Bioremediation Agent	The material must be used within 24-48 hours of constitution of the powdered bacteria portion in water. The dry material must be kept at -20°C and is stable for two years at this temperature. Once sent to the field, the material may be stored on ice for up to two weeks prior to constitution	2
SYSTEM E.T. 20 (formerly MCW.B 20)	Bioremediation Agent	2 years if maintained at 4ºC (39ºF)	2
WASTE AWAY®	Bioremediation Agent	Two years	2
WMI-2000	Bioremediation Agent	2 years if stored at ambient temperatures	2
B&S INDUSTRIAL (see STEP ONE)	Bioremediation Agent	Over 3 years	3
BET BIOPETRO (formerly BET BIOPETRO HEAVY)	Bioremediation Agent	More than 3 years in unopened original shipping container, stored in cool dry area	3
BIOWORLD BIOREMEDIATION HYDROCARBON TREATMENT PRODUCTS	Bioremediation Agent	exceeds 3 years when stored properly in cool, dry place out of direct sun light	3
SHAMANTRA BIO (see SHAMANTRA GREEN)	Bioremediation Agent	3 years when stored in sealed silos, polydrums, poly bags or totes	3
SHAMANTRA GREEN	Bioremediation Agent	3 years when stored in sealed silos, polydrums, poly bags or totes	3
STEP ONE	Bioremediation Agent	Over 3 years	3

VB591™, VB997™, BINUTRIX®	Bioremediation Agent	three years	3
ACT TERRA FIRMA	Bioremediation Agent	If kept in a sealed container with desiccant bag the shelf life is 5 years	5
ACT-TF (see ACT TERRA FIRMA)	Bioremediation Agent	If kept in a sealed container with desiccant bag the shelf life is 5 years	5
DRYLET™ MB BIOREMEDIATION	Bioremediation Agent	Minimum 5 years, with proper storage, in original packaging. Freezing does not harm shelf life; however, extreme heat (over 180°F) for long periods of time can shorten shelf life	5
DUALZORB®	Bioremediation Agent	unopened product is 5 years if not exposed to weather or direct sunlight	5
ERGOFIT MICRO MIX AQUA	Bioremediation Agent	5 years	5
LAND AND SEA RESTORATION	Bioremediation Agent	More than 5 years in unopened original shipping container, store in a cool, dry place	5
MICROSORB SC (see OPPENHEIMER FORMULA)	Bioremediation Agent	5 years	5
OIL SPILL EATER II	Bioremediation Agent	5 years	5
OPPENHEIMER FORMULA	Bioremediation Agent	5 years	5
The OPPENHEIMER FORMULA I (see OPPENHEIMER FORMULA)	Bioremediation Agent	5 year	5
TRAILZORB (see DUALZORB®)	Bioremediation Agent	5 years if not exposed to weather or direct sunlight	5
WHITZORB (see DUALZORB®)	Bioremediation Agent	unopened product is 5 years if not exposed to weather or direct sunlight	5
MICRO-BLAZE®	Bioremediation Agent	Minimum 10 years	10
SUPERSPERSE™ WAO2500	Dispersant	one year	1
MARINE D-BLUE CLEAN™	Dispersant	2 years, if unopened	2
NOKOMIS 3-AA	Dispersant	two years	2
FFT-SOLUTION™	Dispersant	5 years	5
NEOS AB3000	Dispersant	five years	5
MARE CLEAN 200 (formerly MARE CLEAN 505)	Dispersant	10 years when stored indoors	10
NOKOMIS 3-F4	Dispersant	15 years or more if stored correctly in plastic drums	15
ACCELL CLEAN® DWD	Dispersant	unopened drums of ACCELL CLEAN® DWD is unlimited	20

BIODISPERS (formerly PETROBIODISPERS)	Dispersant	unlimited	20
COREXIT® EC9500A (formerly COREXIT 9500)	Dispersant	The shelf life of unopened drums of COREXIT® EC9500A is unlimited	20
COREXIT® EC9500B	Dispersant	The shelf life of unopened drums of COREXIT® EC9500B is potentially unlimited when containers remain capped and sealed to prevent contamination and evaporation of solvents	20
COREXIT® EC9527A (formerly COREXIT 9527)	Dispersant	The shelf life of unopened drums of COREXIT® EC9580A is unlimited	20
DISPERSIT SPC 1000™	Dispersant	The shelf life of Dispersit SPC 1000™ is unlimited in unopened containers	20
FINASOL® OSR 52	Dispersant	unlimited when the containers remain capped and sealed	20
JD-109	Dispersant	unlimited	20
JD-2000™	Dispersant	unlimited	20
SAF-RON GOLD	Dispersant	unlimited in unopened containers	20
SEA BRAT #4	Dispersant	Indefinite when stored properly	20
SEACARE ECOSPERSE 52 (see FINASOL® OSR 52)	Dispersant	potentially unlimited when the containers remain capped and sealed to prevent contamination and evaporation of solvents, and are stored in a place with stable temperatures and are unexposed to direct sunlight	20
SEACARE E.P.A. (see DISPERSIT SPC 1000™)	Dispersant	unlimited in unopened containers	20
SF-GOLD DISPERSANT (see SAF-RON GOLD)	Dispersant	unlimited	20
ZI-400	Dispersant	Unlimited in sealed polydrums of totes	20
ZI-400 OIL SPILL DISPERSANT (see ZI-400)	Dispersant	Unlimited in sealed polydrums of totes	20
MARI-ZYME (see ZYME-FLOW)	Miscellaneous Oil Spill Control Agent – MOSCA	1 year minimum	1
UNITED 658 PETRO-ZYME (see ZYME-FLOW)	Miscellaneous Oil Spill Control Agent – MOSCA	1 year minimum	1
ZYME-FLOW	Miscellaneous Oil Spill Control Agent – MOSCA	1 year minimum	1
ZYME-TREAT (see ZYME-FLOW)	Miscellaneous Oil Spill Control Agent – MOSCA	1 year minimum	1
PX 700™	Miscellaneous Oil Spill Control Agent – MOSCA	Two years	2

NORSOREX® APX	Miscellaneous Oil Spill Control Agent – MOSCA	3 years from date of production	3
ALSOCUP	Miscellaneous Oil Spill Control Agent – MOSCA	five years	5
CIAGENT (formerly CI AGENT, CHEAP INSURANCE, and PETRO-CAPTURE)	Miscellaneous Oil Spill Control Agent – MOSCA	5 years, if stored in cool dry area, away from direct sunlight	5
ELASTOL	Miscellaneous Oil Spill Control Agent – MOSCA	5 years when stored at temperatures below 150ºF	5
LIQUID ELASTOL (see ELASTOL)	Miscellaneous Oil Spill Control Agent – MOSCA	5 years when stored at temperatures below 150ºF	5
OIL BOND®	Miscellaneous Oil Spill Control Agent – MOSCA	5 years	5
OPFLEX®	Miscellaneous Oil Spill Control Agent – MOSCA	5 years	5
SEPARATE (see ELASTOL)	Miscellaneous Oil Spill Control Agent – MOSCA	5 years	5
PES-51	Miscellaneous Oil Spill Control Agent – MOSCA	6 years (unopened drum), 1 year (opened drum)	6
GELCO 200	Miscellaneous Oil Spill Control Agent – MOSCA	10 years when stored in dry, cool area out of sunlight	10
AQUA N-CAP™ POLYMER (see OIL SOLUTIONS POWDER)	Miscellaneous Oil Spill Control Agent – MOSCA	Not Limited	20
OIL SOLUTIONS POWDER	Miscellaneous Oil Spill Control Agent – MOSCA	Not limited	20
RAPIDGRAB 2000™	Miscellaneous Oil Spill Control Agent – MOSCA	unlimited	20
WASTE-SET #3200®	Miscellaneous Oil Spill Control Agent – MOSCA	unlimited	20
WASTE-SET #3400®	Miscellaneous Oil Spill Control Agent – MOSCA	unlimited	20
SILTECH OP-40	Surface Collecting Agent	three years	3
THICKSLICK 6535	Surface Collecting Agent	three years	3
CLEAN SPLIT (see SPLIT DECISION SC)	Surface Washing Agent	one year	1

DUO-SPLIT (see SPLIT DECISION SC)	Surface Washing Agent	one year	1
GREEN TECHNOLOGIES SOLUTIONS-OIL RECOVERY (GTS-OR)	Surface Washing Agent	One year minimum when stored between 50°F and 100°F	1
SPLIT DECISION SC (formerly SPLIT DECISION)	Surface Washing Agent	one year	1
TXCHEM HE-1000™	Surface Washing Agent	One year in a sealed container	1
AQUACLEAN	Surface Washing Agent	18 months if stored between 50ºF and 104ºF, and away from acids	1.5
BIOGRASS® EXTRA	Surface Washing Agent	Two years minimum	2
CN-110	Surface Washing Agent	2 years	2
CORIBA 700 ER (see CORIBA 700 SR)	Surface Washing Agent	Two years unopened minimum	2
CORIBA 700 OS (see CORIBA 700 SR)	Surface Washing Agent	Two years unopened minimum	2
CORIBA 700 SR	Surface Washing Agent	Two years unopened minimum	2
CORIBA 713 ER (see CORIBA 713 SR)	Surface Washing Agent	Two years unopened minimum	2
CORIBA 713 OS (see CORIBA 713 SR)	Surface Washing Agent	Two years unopened minimum	2
CORIBA 713 SR	Surface Washing Agent	Two years unopened minimum	2
DE-SOLV-IT CLEAN AWAY APC SUPER CONCENTRATE	Surface Washing Agent	Two years in sealed drums or totes	2
DE-SOLV-IT INDUSTRIAL FORMULA	Surface Washing Agent	Two years in sealed polydrums or totes	2
DO-ALL #18	Surface Washing Agent	at least two years	2
ECOVOOM-MARINE (see JEP-MARINE CLEAN)	Surface Washing Agent	minimum 2 years	2
ETHOS CLEAN	Surface Washing Agent	2 years in a sealed (unopened) container (tote, drum, pail) as delivered. 1 year in an opened container).	2
JEP-MARINE CLEAN	Surface Washing Agent	minimum 2 years	2
MARINE GREEN CLEAN™	Surface Washing Agent	2 years	2
MARINE GREEN CLEAN PLUS™	Surface Washing Agent	2 years	2

NONTOX™	Surface Washing Agent	minimum of two years	2
OSR-10	Surface Washing Agent	2 years in a sealed (unopened) container (tote, drum, pail)	2
PROCLEANS	Surface Washing Agent	2 years at recommended temperatures if unopened	2
SOC 10	Surface Washing Agent	two years	2
SUPERALL #38 (see TOPSALL #30)	Surface Washing Agent	two years	2
TOPSALL #30	Surface Washing Agent	two years	2
TULXA	Surface Washing Agent	Two years if stored in dry, fresh environment	2
EO ALL PURPOSE SOAP-LAVENDER	Surface Washing Agent	Three years	3
FIREMAN'S BRAND SPILLCLEAN (see SPILLCLEAN)	Surface Washing Agent	3 years. Avoid extreme heat and store in a dry, cool area	3
SPILLCLEAN or SPILLCLEAN ["Concentrate"]	Surface Washing Agent	three years	3
CLEAN GREEN	Surface Washing Agent	5 years in sealed polydrums or totes	5
CLEANGREEN® PLANET WASH (see CLEAN GREEN)	Surface Washing Agent	5 years in sealed polydrums or totes	5
MICRO CLEAN (see NATURE'S WAY HS)	Surface Washing Agent	five years	5
NATURE'S WAY HS	Surface Washing Agent	five years	5
NATURE'S WAY PC (see NATURE'S WAY HS)	Surface Washing Agent	five years	5
POWERCLEAN (see NATURE'S WAY HS)	Surface Washing Agent	five years	5
PREMIER 99	Surface Washing Agent	5 years	5
SAFE KLEEN	Surface Washing Agent	Five years if stored in tightly closed containers under dry conditions within temperature range	5
SC-1000™	Surface Washing Agent	Minimum of 5 years	5
SIMPLE GREEN®2013 Reformulation	Surface Washing Agent	five years	5
G-CLEAN OSC-1809	Surface Washing Agent	Five to ten years in sealed polydrums or totes	7.5
OIL SPILL CLEANUP (see G-CLEAN OSC-1809)	Surface Washing Agent	Five to ten years in sealed polydrums or totes	7.5
DYNAMIC GREEN™	Surface Washing Agent	6-10 years in sealed poly drums or totes	8

BIOSOLVE® HYDROCARBON MITIGATION™ AGENT	Surface Washing Agent	10+ year shelf life if unopened	10
CYTOSOL	Surface Washing Agent	Closed container: 10 years in a dry environment. Open container: 1 year in a warm, humid environment. The product does not deteriorate appreciably over time, but will grow bacteria if water condensation accumulates in the container.	10
F-500	Surface Washing Agent	15 years when stored between 35ºF - 130ºF in unopened containers	15
NOKOMIS 5-W	Surface Washing Agent	15 years or more if stored correctly in plastic drums	15
ACCELL CLEAN® SWA	Surface Washing Agent	unopened drums of ACCELL CLEAN® SWA is unlimited	20
ALL PURPOSE CLEANER & REMEDIATOR (see GREEN BEAST™ OIL SPILL & ODOR REMEDIATOR)	Surface Washing Agent	Unlimited in sealed polydrums or totes	20
AWAN PRA OIL FIELD SOLUTION™ (see EPA OIL FIELD SOLUTION™)	Surface Washing Agent	Unlimited in sealed polydrums or totes (as delivered). Avoid direct sunlight.	20
BG-CLEAN™ 401	Surface Washing Agent	unlimited	20
COREXIT® EC9580A (formerly COREXIT 9580 SHORELINE CLEANER)	Surface Washing Agent	unlimited	20
ENVIROCLEAN (formerly ENVIRO CLEAN 165)	Surface Washing Agent	Unlimited if unopened	20
ENVIRONMENTAL 1 CRUDE OIL CLEANER	Surface Washing Agent	Unlimited in sealed polydrums or totes	20
ENVIRONMENTAL 1 WASHING AGENT (see ENVIRONMENTAL 1 CRUDE OIL CLEANER)	Surface Washing Agent	Unlimited in sealed polydrums or totes	20
EPA OIL FIELD SOLUTION™	Surface Washing Agent	Unlimited in sealed polydrums or totes (as delivered). Avoid direct sunlight.	20
E-SAFE©	Surface Washing Agent	Unlimited if left in unopened containers stored at 40º - 110ºF and away from direct sunlight	20
GLOBAL ENVIRONMENTAL CLEANER™ (see EPA OIL FIELD SOLUTION™)	Surface Washing Agent	Unlimited in sealed polydrums or totes	20
GOLD CREW SW	Surface Washing Agent	20 years (unopened)	20

GREEN BEAST OIL SPILL & ODOR REMEDIATOR	Surface Washing Agent	Unlimited in sealed polydrums or totes	20
GREEN BEAST WASHING AGENT (see GREEN BEAST OIL SPILL & ODOR REMEDIATOR	Surface Washing Agent	Unlimited in sealed polydrums or totes	20
HYDRO-CLEAN™ (see EPA OIL FIELD SOLUTION™)	Surface Washing Agent	Unlimited in sealed polydrums or totes	20
NALE-IT	Surface Washing Agent	Indefinite	20
NATURAMA G3 A-5	Surface Washing Agent	unlimited	20
PETRO-CLEAN	Surface Washing Agent	Indefinite when stored properly	20
PETRO-GREEN ADP-7	Surface Washing Agent	indefinite	20
PETROMAX PSC 3	Surface Washing Agent	Unlimited when stored in unopened, sealed 5 gallon containers, 55 gallon drums, polydrums, and totes	20
PETROMAX SOIL CLEANING AND WASHING AGENT (see PETROMAX PSC 3)	Surface Washing Agent	Unlimited when stored in unopened, sealed 5 gallon containers, 55 gallon drums, polydrums, and totes	20
PETROTECH 25	Surface Washing Agent	unlimited shelf-life when maintained in the factory sealed containers and stored within the prescribed temperature limits	20
SANDKLENE 950	Surface Washing Agent	Unlimited in sealed drums or totes	20
SHEEN-MAGIC©	Surface Washing Agent	Unlimited if left in unopened containers stored at 40º - 110ºF and away from direct sunlight	20
SIMPLE GREEN®	Surface Washing Agent	unlimited	20
VERU-SOLVE™ MARINE 200 HP	Surface Washing Agent	Indefinite, product will not degrade over time	20

To: Wilson, Gregory[Wilson.Gregory@epa.gov]
From: Principe, Vanessa
Sent: Thur 5/7/2015 7:46:02 PM
Subject: Emailing: Consultation_BO_Alaska Unified Plan_Attachment_DOI Sub J Comment_Letter.pdf
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