LWG
LOWER WILLAMETTE GROUP

PORTLAND HARBOR RI/FS

COMPREHENSIVE ROUND 2 SITE CHARACTERIZATION SUMMARY AND DATA GAPS ANALYSIS REPORT

APPENDIX G
ROUND 2 ECOLOGICAL RISK ASSESSMENT

February 21, 2007

PORTLAND HARBOR SUPERFUND SITE
COMPREHENSIVE ROUND 2 SITE CHARACTERIZATION
SUMMARY AND DATA GAPS ANALYSIS REPORT

APPENDIX G: ECOLOGICAL RISK ASSESSMENT

February 21, 2007

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LIST OF ACRONYMS

ACR	acute-chronic ratio
Avocet	Avocet Consulting
AWQC	ambient water quality criteria
BCF	bioconcentration factor
BEHP	bis(2-ethylhexyl) phthalate
BERA	baseline ecological risk assessment
BMF	biomagnifications factor
BSAF	biota-sediment accumulation factor
bw	body weight
CBFWA	Columbia Basin Fish and Wildlife Authority
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
COC	chemical of concern
COI	chemical of interest
COPC	chemical of potential concern
CSM	conceptual site model
dw	dry weight
EC50	concentration that causes a non-lethal effect in 50% of an exposed population
Eco SL	ecological screening level
Eco SSL	ecological soil screening level
EPA	US Environmental Protection Agency
EPC	exposure point concentration
EqP	equilibrium partitioning
ERA	ecological risk assessment
ERAGS	Ecological Risk Assessment Guidance for Superfund
EROD	ethoxyresorufin-O-deethylase
ESA	Endangered Species Act
ESB	equilibrium partitioning sediment benchmark
FACR	final acute-chronic ratio
FAV	final acute value
FCV	final chronic value
FIR	food ingestion rate
FPM	floating percentile model
FSP	field sampling plan
FWM	food web model
GMAV	genus mean acute value
HCH	hexachlorocyclohexane

HHRA	human health risk assessment
HPAH	high-molecular-weight polycyclic aromatic hydrocarbon
HQ	hazard quotient
iAOPC	initial area of potential concern
iCOC	initial chemical of concern
Integral	Integral Consulting, Inc.
iPRG	initial preliminary remediation goal
IP	intraperitoneal
ISA	Initial Study Area
LC50	lethal dose that causes the death of 50% of an exposed population
LOAEL	lowest-observed-adverse-effect level
LOE	line of evidence
LPAH	low-molecular-weight polycyclic aromatic hydrocarbon
LRM	logistic regression model
LWG	Lower Willamette Group
LWR	Lower Willamette River
MDD	minimum detectable difference
NAPL	non-aqueous phase liquid
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no-observed-adverse-effect level
NTP	National Toxicity Program
O&M	operation and maintenance
OAR	Oregon Administrative Rules
ODEQ	Oregon Department of Environmental Quality
ODFW	Oregon Department of Fish and Wildlife
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PEL	probable effects level
POTW	publicly owned treatment works
PRE	preliminary risk evaluation
RI/FS	remedial investigation/feasibility study
RM	river mile
SCRA	site characterization and risk assessment
SAV	secondary acute value
SCV	secondary chronic value
SEA	Striplin Environmental Associates, Inc.
SEL	severe effects level
SI	sediment ingestion
SIR	sediment ingestion rate

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SLERA	screening-level ecological risk assessment
SMAV	species mean acute value
SUF	site use factor
SQV	sediment quality value
SVOC	semivolatile organic compound
TBT	tributyltin
TCDD	tetrachlorodibenzo-*p*-dioxin
TEF	toxic equivalency factor
TEL	threshold effects level
TEQ	toxic equivalent
TOC	total organic carbon
TPH	total petroleum hydrocarbons
TRV	toxicity reference value
TZW	transition zone water
UCL	upper confidence limit
UF	uncertainty factor
USFWS	US Fish and Wildlife Service
USGS	US Geological Survey
VOC	volatile organic compound
WQC	water quality criteria
WHO	World Health Organization
Windward	Windward Environmental LLC
ww	wet weight
XAD	Infiltrex system with XAD resin column

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1.0 INTRODUCTION

This document presents the ecological risk assessment (ERA) component of the *Portland Harbor Superfund Site Comprehensive Round 2 Site Characterization Summary and Data Gaps Analysis Report*, hereafter referred to as the Comprehensive Round 2 Report.

A remedial investigation/feasibility study (RI/FS) is designed to be an iterative process that addresses the relationships among the factors that may affect chemical distribution, risk estimates, and remedy selection. Currently, two major rounds of field investigations have been completed as part of the overall RI/FS. Round 1 was conducted in 2002 and focused primarily on chemical concentrations in fish and shellfish tissue and in beach sediments. Round 2 was conducted in 2004 and 2005 and focused on chemical concentrations in sediment core samples, in-water surface sediments, surface water, transition zone water (TZW), and additional shellfish tissue and beach sediment. These Round 1 and Round 2 sampling efforts, while focused on River Mile (RM) 3.5 to RM 9.2, the length of the Initial Study Area (ISA) as defined by the Administrative Order on Consent (EPA 2001), extended well beyond the ISA to RM 2.0 downstream and to RM 11 upstream. This 9- mile portion of Portland Harbor, RM 2.0 to RM 11, is referred to as the Study Area.

This Round 2 ERA is being conducted as part of the Comprehensive Round 2 Report for purposes of identifying ecological initial chemicals of concern (iCOCs) the RI/FS data needs for Round 3 to complete a baseline ERA (BERA). A BERA will be conducted after the completion of Round 3 sampling. The results of the BERA will be used in developing remedial action objectives.

1.1 OBJECTIVES

The general objective of an ERA is to characterize the potential risks to ecological receptors that may be posed by chemicals present in or entering into environmental media (i.e., water, air, or soil) or bioaccumulating in the food chain. The overall purpose of this Round 2 ERA for the Portland Harbor Superfund Site is to determine which chemicals have the potential to cause or increase the risk to ecological receptors, provide a basis for identifying initial preliminary remediation goals (iPRGs) and initial areas of potential concern (iAOPCs), and identify additional information needed in order to adequately conduct a BERA of Portland Harbor. This Round 2 ERA also provides the basis for updating the preliminary problem formulation, including the conceptual site model (CSM).

The specific objectives of the Comprehensive Round 2 Report ERA are as follows:

- Perform and document a conservative screening-level risk assessment of chemicals of interest (COIs) using site data collected to date, and develop a list of Round 2 chemicals of

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potential concern (COPCs) based on the conservative screening-level risk assessment assumptions for all receptor groups

- Perform and document further screening of the Round 2 COPCs using site data collected to date and more realistic risk assessment assumptions and develop a list of iCOCs based on this additional evaluation[1]
- Identify any remaining data needs to be filled prior to the preparation of the BERA

The results of this ERA were used in the Comprehensive Round 2 Report to do the following:

- Develop iPRGs and iAOPCs
- Update the preliminary problem formulation, including the CSM

1.2 APPROACH

This Round 2 ERA represents an interim phase of the BERA for the Study Area and presents preliminary risk analyses, including documentation of the screening-level risk assessment, per US Environmental Protection Agency (EPA) 1997 guidelines (EPA 1997a) and is consistent with EPA 1998 guidance (EPA 1998). For several receptors (e.g., wildlife), this Round 2 evaluation is an update to the draft Portland Harbor ecological preliminary risk evaluation (PRE), hereafter referred to as the Ecological PRE, which was delivered to EPA on September 9, 2005 (Windward 2005).

This Round 2 ERA follows the methods and outline that were included in the following documents:

- Administrative Order on Consent (EPA 2001)
- Appendix B: Ecological Risk Assessment Approach of the *Portland Harbor RI/FS Programmatic Work Plan* (Integral et al. 2004a), which was approved by EPA in June 2004.
- *Portland Harbor Superfund Site Ecological Risk Assessment: Comprehensive Synopsis of Approaches and Methods* (Windward 2004b), submitted to EPA on June 28, 2004. EPA provided comments to this document on October 1, 2004 (EPA 2004a). The Lower Willamette Group (LWG) has incorporated EPA's comments in this ERA.

[1] The identification of chemicals of concern (COCs) is beyond the scope of the Comprehensive Round 2 Report ERA. COCs will be identified in the BERA as part of the Remedial Investigation (RI).

- Portland Harbor Superfund Site Ecological Risk Assessment: Estimating Risks to Benthic Organisms Using Predictive Models Based on Sediment Toxicity Tests (Windward et al. 2006), submitted to EPA on March 17, 2006. EPA provided comments to this document on July 6, 2006 (EPA 2006c). LWG has incorporated EPA's comments on this document in this ERA, as documented in LWG's response to EPA comments submitted on September 1, 2006 (LWG 2006b).
- *Portland Harbor RI/FS Ecological Preliminary Risk Evaluation* (Windward 2005), submitted to EPA on September 9, 2005. EPA provided comments on this document on April 28, 2006 (EPA 2006b), and subsequent clarification comments on toxicity reference values (TRVs) on July 6, 2006 (EPA 2006e), and on August 24, 2006 (EPA 2006a). LWG has incorporated EPA's comments on the Ecological PRE and subsequent comments on the selection of TRVs in this ERA.
- *Portland Harbor Superfund Site Proposed Ecological Risk Assessment Decision Framework,* submitted to EPA on March 15, 2006 (Windward 2006b). Following the submittal of this document, EPA developed a framework table outlining the ecological measurement endpoints and lines of evidence (LOEs) for evaluating ecological risks, which was provided to LWG on August 24, 2006 (EPA 2006a).[2] LWG has agreed to evaluate all of the LOEs for assessing risk to ecological receptors (EPA 2006a).

1.3 ORGANIZATION

This Round 2 ERA incorporates the four steps of the BERA process: data collection and evaluation, exposure assessment, toxicity assessment, and risk characterization. The document is arranged as follows:

- Section 2.0 – Data Evaluation
- Section 3.0 – Benthic Invertebrate Risk Assessment
- Section 4.0 – Fish Risk Assessment
- Section 5.0 – Wildlife Risk Assessment
- Section 6.0 – Amphibian and Reptile Risk Assessment
- Section 7.0 – Aquatic Plants Risk Assessment
- Section 8.0 – Summary and Data Gaps Analysis
- Section 9.0 – References

[2] EPA's measurement endpoint and LOE table is presented in Sections 3.0 through 7.0, as appropriate, for each receptor group.

- Attachment G1 – Data Management
- Attachment G2 – Screening Assessment for Benthic Invertebrates
- Attachment G3 – Ecological Screening Levels
- Attachment G4 – Screening Assessment for Fish
- Attachment G5 – Toxicity Reference Value Round 2 Toxicity Reference Value Addendum for Fish and Wildlife
- Attachment G6 – Screening Assessment for Wildlife
- Attachment G7 – Screening Assessment Amphibians and Reptiles
- Attachment G8 – Screening Assessment for Aquatic Plants

2.0 ERA DATASET

This section presents the data that were used in the Round 2 ERA and a summary of the data evaluation process used for all receptor groups (i.e., benthic invertebrates, fish, wildlife, amphibians and reptiles, and aquatic plants). Details on selected ecological receptors of concern are presented in Sections 3.0 through 7.0. The LWG sampling events and non-LWG sampling events included in the site characterization and risk assessment (SCRA) dataset are described in detail in Section 2.0 of the Comprehensive Round 2 Report. The dataset used in this ERA and described in this section (hereafter referred to as the ERA dataset) is a subset of the data that comprise the SCRA dataset.

2.1 SUMMARY OF ERA DATA

The ERA dataset includes only those matrices relevant for ecological exposure pathways: surface (0 to 30 cm) sediment, benthic and fish tissue, surface water, and shallow (0 to 38 cm) TZW. Other matrices included in the SCRA dataset (e.g., subsurface sediment and deep [90 to 150 cm] TZW samples) were not used in this ERA. The ERA dataset is summarized, by matrix, in Table 2-1 and briefly described in the following subsections. Detailed information on the sampling events is presented in Section 2.0 of the Comprehensive Round 2 Report. The data management rules for the ERA dataset are presented in Attachment G1.

2.1.1 Surface Sediment

Surface sediment chemistry in the ERA dataset includes LWG-collected data (from various sampling events in Rounds 1 and 2) and non-LWG collected data.[3] Table 2-2 presents a summary of the surface sediment samples included in the ERA dataset. All non-LWG data included in the SCRA database (see Section 2.0 of the Comprehensive Round 2 Report) were of acceptable data quality for risk evaluation (Category 1/QA2), as agreed to between LWG, EPA, and EPA's partners in the programmatic work plan (Integral et al. 2004a).

All surface sediment data included in the ERA dataset were collected from within the top 30.5 cm of the sediment horizon and located within the Study Area.[4] Sediment natural attenuation cores collected by LWG for nature and extent were not included in the ERA dataset because multiple depth intervals in small increments (as small as 4 cm) were collected within the 0 to 30.5 cm surface sediment depth horizon, and these cores were collected to support the nature and extent evaluation. Surface sediment samples that were collected from areas that have since been dredged were not

[3] Any additional changes to the SCRA dataset since June 2006 are not reflected in this ERA dataset (e.g., samples that have since been dredged).
[4] The Study Area is defined as river mile (RM) 2 to 11. For purposes of the ERA dataset, samples collected between RM 1.91 and RM 11 were considered part of the Study Area.

included in the SCRA and ERA datasets because these samples no longer represent the current conditions at the Study Area.

Surface sediment data were used to estimate exposure concentrations for relevant ecological receptors based on direct contact (i.e., benthic invertebrates, fish, and aquatic plants) and dietary exposure (i.e., fish and wildlife). The chemistry results of surface sediment samples from these sampling events are presented in Section 2.0 of the Comprehensive Round 2 Report.

2.1.2 Benthic Invertebrate and Whole-Body Fish Tissue

Tissue chemistry data used for the ERA dataset were selected from various LWG sampling events, to evaluate potential risks to selected ecological receptors of concern. Table 2-3 presents a summary of the fish and benthic invertebrate tissue samples included in the ERA dataset. All tissue data included in the ERA dataset were collected from within the Study Area. Benthic invertebrate tissue in the ERA dataset included field-collected tissue samples for crayfish,[5] clam (*Corbicula* spp.), and epibenthic invertebrates and zooplankton collected with multiplate samplers. Tissue samples of clams (*Corbicula fluminea*) and worms (*Lumbriculus variegatus*) following bioaccumulative testing in the laboratory from surface sediments collected within the Study Area were also included in the ERA dataset.

All fish tissue data were collected in the field and are based on whole-body tissue composites. Fish tissue composites from the site were analyzed for nine fish species: largescale sucker (*Catostomus macrocheilus*), carp (*Cyprinus carpio*), juvenile chinook salmon (*Oncorhynchus tshawytscha*), sculpin *Cottus* spp.), peamouth (*Mylocheilus caurinus*), smallmouth bass (*Micropterus dolomieui*), northern pikeminnow (*Ptychocheilus oregonensis*), black crappie (*Pomoxis nigromaculatus*), and brown bullhead (*Ameiurus nebulosus*). Results of the chemical analysis of juvenile chinook salmon stomach content samples from within the Study Area were also included in the ERA dataset. Several fish species composites were based on individual fish collected over various stretches of the river within the Study Area, as detailed below.

- Smallmouth bass tissue composites were based on individual fish collected over 1-mile stretches.
- Largescale sucker, peamouth, northern pikeminnow tissue composites were based on individual fish collected over 2-mile stretches.
- Black crappie, brown bullhead, and carp tissue composites were based on individual fish collected over 3-mile stretches.

[5] Crayfish were not identified to species; however, only two species, the western freshwater crayfish (*Pacifastacus leniusculus*) and the Siberian prawn (*Exopalaemon modestus*), occur in the Study Area.

Fish and invertebrate tissue data were used to estimate exposure concentrations for relevant pathways (e.g., dietary exposure) and ecological receptors (i.e., benthic invertebrates, fish, and wildlife). The chemistry results of tissue samples from these sampling events are presented in Section 2.0 of the Comprehensive Round 2 Report.

2.1.3 Surface Water

Surface water data used in the ERA dataset includes all Round 2 LWG-collected data. During Round 2, surface water sampling was performed in three separate events in November 2004, March 2005, and July 2005 to capture the seasonal water flow levels on the Lower Willamette River (LWR). The November 2004 event was conducted during mid- to late fall to capture potentially elevated chemical concentrations in the river from rainfall runoff. The March 2005 event was selected by EPA to coincide with the early exposure period for amphibian egg masses. The July 2005 event was timed to coincide with low-flow conditions, when any effects of groundwater discharge to the water column would be most pronounced. Additional surface water sampling is anticipated as part of Round 3 sampling and will be included in the BERA. Table 2-4 presents a summary of the surface water samples included in the ERA dataset.

Twenty-three surface water locations were sampled within the Study Area during each sampling event, and surface water samples were either near-bottom samples (i.e., collected within 1 to 3 ft of the river bottom) or integrated water column samples. Data from all three sampling events were included in the ERA dataset. Surface water samples were collected from 14 amphibian habitat locations, 3 beach locations, 3 human-use areas, and 3 transect locations (at RM 4.0, RM 6.3, and RM 11). Several types of surface water samples were collected, including single-point near-bottom samples, single-point water column samples, and cross-sectional river transect water column samples. Surface water samples were collected using two types of sampling methods: 1) a peristaltic pump, or 2) an Infiltrex™ 300 system with an XAD-2 resin column (XAD). Surface water samples collected using the peristaltic pump were collected using a standard volume (i.e., 10 to 20 L) over a short period of time (i.e., 30 minutes) and were analyzed for standard and conventional parameters (e.g., metals, pesticides, herbicides, and SVOCs). Surface water samples for hydrophobic organic compounds (e.g., dioxins and furans, PCBs, PAHs, pesticides, and phthalates) were collected using an XAD system for a high volume (i.e., 1,000 L) at a rate of 1.25 L per minute over approximately13 hours. The XAD system also incorporated a glass fiber filter to capture the particulate fraction of the water and used an ultra-low analytical detection limit to analyze the hydrophobic organic compounds. The filter and the column analytical results were combined to determine total analyte concentrations in the water column.

Surface water data were used to estimate exposure concentrations via surface water for relevant ecological receptors (i.e., benthic invertebrates, fish, amphibians, and aquatic plants). The chemistry results of surface water samples from these sampling events are presented in Section 2.0 of the Comprehensive Round 2 Report.

2.1.4 Transition Zone Water

TZW data included in the ERA dataset includes all shallow (0 to 38 cm) data collected by LWG in Round 2. During Round 2 sampling, TZW sampling was performed between October 3 and December 2, 2005, to capture the groundwater discharge to the LWR. Table 2-5 presents a summary of the shallow (0 to 38 cm) TZW samples included in the ERA dataset.

The TZW sampling locations selected for each site focused primarily on the zones of possible groundwater plume discharge, based on the Round 2 groundwater pathway assessment pilot study discharge mapping effort conducted from August 1 to September 9, 2005 (Integral 2006). Nine high-priority Category A sites, defined as sites with a confirmed or reasonable likelihood for discharge of upland groundwater chemicals of interest (COIs) to Portland Harbor were selected as TZW locations and sampled within the Study Area (RM 2.0 to RM 11). TZW samples were collected from the following nine sites: Kinder Morgan Linnton Terminal, Arco Terminal 22T, ExxonMobil Oil terminal, Gasco, Siltronic, Rhône-Poulenc, Arkema (acid plant and chlorate plant areas), Willbridge bulk fuels terminal, and Gunderson. TZW samples were collected with either a Trident® (30-cm) probe, Geoprobe®, or small-volume peeper. TZW samples were collected at depths up to 150 cm; however, the TZW data in the ERA dataset included only samples from the biologically active zone (≤ 38 cm), where exposure of receptors could occur. The Trident is a direct-push system that is equipped with temperature, conductivity, and water sampling probes. With the Trident probe, TZW is collected through a small-diameter, Teflon®-coated, stainless steel probe with a port on the end covered by steel mesh. The Geoprobe® used is comparable to the Trident probe method. Peepers used to sample TZW were 6- by 18-in. plate peepers with 28 rows of sample ports and a 5-µm membrane. All of the peepers were deployed by a diver and were left in place to equilibrate for a 3-week period. After the equilibration period, the peepers were retrieved, and samples were collected by inserting a needle through the membrane to extract water (Integral 2006).

TZW data were used to estimate TZW exposure concentrations for relevant ecological receptors (i.e., benthic invertebrates, fish, and aquatic plants). The chemistry results of TZW samples from this sampling event are presented in Section 2.0 of the Comprehensive Round 2 Report.

2.2 ROUND 2 ERA DATA EVALUATION PROCESS

This section describes the process used in the risk assessment to narrow the list of chemicals that might pose risks to ecological receptors. iCOCs were identified through a series of screening steps, quantitative evaluations, and qualitative evaluations (Figure 2-1). COIs were identified as any chemical detected in the dataset. Seven crustal elements were evaluated in a pre-screening step (see Section 2.2.2) to determine if they would be carried forward in the risk assessment process. Those chemicals whose maximum concentrations exceeded the screening criteria were then maintained as Round 2 COPCs. COPCs were further assessed through one

or more steps to identify iCOCs, which may potentially pose a risk to ecological receptors.

2.2.1 Evaluation of Crustal Elements

Ecological COIs were identified as all chemicals detected in various media (i.e., tissue, surface sediment, surface water, or TZW). However, several of these COIs (i.e., aluminum, beryllium, cobalt, iron, magnesium, manganese and potassium) are naturally occurring crustal elements, so a pre-screening step was conducted to assess whether they should be carried forward in the risk assessment process. Aluminum and manganese were detected in tissue, sediment, and water. Regional aluminum and manganese concentrations in soils are consistent with the concentrations of sediment in the Study Area. In Clark County, Washington, just across the Columbia River border with Oregon, aluminum and manganese levels are 52,300 and 1,500 mg/kg, respectively (Ecology 1994). Surface sediment concentrations of aluminum in the Study Area ranged from 1,630 to 46,200 mg/kg. Surface sediment manganese concentrations ranged from 265 to 2,130 mg/kg, with almost all samples (99.5%) less than 1,500 mg/kg.

Beryllium, cobalt, iron, magnesium, and potassium were detected in TZW and surface sediment but were not analyzed in tissue and surface water. Regional background concentrations were available for beryllium, iron, and magnesium. All surface sediment concentrations of beryllium in the Study Area (ranging from 0.217 to 0.90 mg/kg) were less than the reported Clark County, Washington, background value of 2.0 mg/kg (Ecology 1994), and almost all surface sediment samples (99%) had magnesium concentrations less than the average concentration (7,890 mg/kg) reported by the US Geological Survey (USGS) in Clark County, Washington (USGS 2006). Iron concentrations in surface sediment samples in the Study Area ranged from 29,200 to 64,500 mg/kg. Eighty-four percent exceeded the Clark County background concentration (36,100 mg/kg), but all samples were within the soil background iron concentration range (100 to 100,000 mg/kg) reported by the National Oceanic and Atmospheric Administration (NOAA) (1999). No regional background concentrations of potassium and cobalt were found. However, potassium is a naturally occurring crustal element found in igneous rocks, and cobalt is associated with basalt, a common type of rock found in the LWR.

None of these crustal elements is expected to pose risks to ecological receptors because concentrations in the Study Area were at or below regional background levels, or these elements are ubiquitous in environmental media because they occur naturally. Therefore, none of these metals (i.e., aluminum, beryllium, cobalt, iron, magnesium, manganese and potassium) was evaluated as a COI, and no further analysis was conducted. Sediment concentrations of these metals were plotted by RM to depict the spatial distribution of the metals throughout the Study Area. The plots of aluminum, beryllium, cobalt, iron, magnesium, manganese, and potassium are uniform across the Study Area and indicate that the concentrations of these COIs in sediment are natural and not the result of Comprehensive Environmental Response,

Compensation, and Liability Act (CERCLA) site activities (see Figures 2-2 through 2-8).

2.2.2 Identification of Round 2 COPCs

Following the prescreening of crustal elements, all remaining COIs were screened using the ERA dataset and conservative risk assessment assumptions to yield the list of Round 2 COPCs. A screening-level ecological risk assessment (SLERA) was conducted for all receptor groups to identify Round 2 COPCs. The weight-of-evidence approach for Round 2 COPC identification was as follows:

- Round 2 COPCs were identified for each LOE.[6]
- Round 2 COPC lists were integrated across LOEs to determine the list of Round 2 COPCs for each receptor group (i.e., benthic invertebrates, fish, wildlife, amphibians/reptiles, and aquatic plants).

In the screening-level approach, any chemical that screened in based on any one LOE was carried forward as a Round 2 COPC. All LOEs for all receptors (identified in Sections 3 through 7) were used to identify Round 2 COPCs, with the exception of the SQV comparison LOE for benthic invertebrates, which was given a weight of zero because none of the SQVs in the literature could reliably predict toxicity to the benthic community in the Study Area based on 233 sediment toxicity tests.

The process for identifying Round 2 COPCs was consistent across all LOEs and all receptor groups, with the exception of the sediment toxicity LOEs for benthic invertebrates. For the sediment toxicity LOE, Round 2 COPCs were identified as those chemicals associated with observed toxicity based on the predictive sediment toxicity models (FPM and LRM). For all LOEs other than the sediment toxicity LOEs, the maximum concentration of each COI (or maximum estimated exposure dose, for the dietary-dose LOEs) was compared to its respective screening-level toxicological threshold (e.g., Eco SL, TRV). If the maximum exposure concentration or exposure dose (for each LOE) was greater than the screening-level toxicological threshold, the chemical was identified as a Round 2 COPC. The identification of Round 2 COPCs by means of a conservative screen in which maximum concentrations in relevant media are used is consistent with EPA guidance (EPA 1997a, b) and an important step in narrowing the list of chemicals to only those chemicals that could potentially pose a risk to ecological receptors.

Detailed COI screening methods and results are presented in Attachments G2, G4, G6, G7, and G8 for benthic invertebrates, fish, wildlife, amphibians/reptiles, and aquatic plants.

[6] The SQV comparison LOE for benthic invertebrates was the only LOE not used to identify Round 2 COPCs.

2.2.3 Identification of iCOCs

A further assessment of Round 2 COPCs was conducted for all receptor groups to identify iCOCs that may potentially pose a risk to ecological receptors. iCOC identification relied on the same approach as that used for Round 2 COPC identification, differing only in level of conservatism. The iCOC identification used less-conservative assumptions and incorporated a weighted approach across LOEs in an effort to narrow the list of chemicals and identify the potential areas associated with risks to ecological receptors in the Study Area. Round 2 COPC screening methods and results are presented by receptor group in Sections 3.0 to 7.0.

3.0 BENTHIC INVERTEBRATE RISK ASSESSMENT

This section presents the Round 2 ERA for benthic invertebrates in the Portland Harbor Study Area. The problem formulation, including the results of the screening assessment to identify benthic Round 2 COPCs, is presented in Section 3.1. Round 2 COPCs for each line of evidence were identified using data collected to date based on screening-level evaluations for each line of evidence. The different measures of exposure and effects are presented in the following sections: sediment toxicity testing and modeling (Section 3.2), evaluation of tissue chemistry data (Section 3.3), and evaluation of exposure to near-bottom surface water (Section 3.4) and TZW (Section 3.5). The preliminary risk characterization is presented in Section 3.6, and the uncertainties associated with the preliminary risk characterization are discussed in Section 3.7. The Round 2 risk conclusions that will be used to inform the BERA are presented in Section 3.8.

3.1 PROBLEM FORMULATION

The benthic resources within the Study Area that are potentially at risk based on site-specific data collected during Rounds 1 and 2 are presented in Section 3.1.1. The selection of benthic receptors and the rationale for the selections are presented in Section 3.1.2. The benthic CSM, which describes the different exposure media, pathways, and their significance in exposure of benthic invertebrates, is presented in Section 3.1.3. The ecological measurement endpoints and LOEs for evaluating risks to benthic invertebrates are presented in Section 3.1.4.

3.1.1 Benthic Invertebrate Resources Potentially at Risk

Habitat utilization and potential benthic invertebrate receptors at risk were identified and described in detail in Appendix B of the Programmatic Work Plan (Integral et al. 2004b). This section presents a summary of potential benthic resources at risk based on site-specific data collected during Rounds 1 and 2.

Benthic invertebrates serve various functions in large river ecosystems. Infaunal and epifaunal invertebrates often constitute a significant portion of the heterotrophic biomass in a river system (Jahn and Anderson 1986) and therefore serve as a principal food resource for higher-trophic-level consumers. Invertebrates also control energy flow by acting as a principal processor of organic matter (Merritt et al. 1984).

The Study Area is characterized by a navigation channel (which is maintained through active dredging) with a developed shoreline (e.g., rock riprap, sheet pile, bulkheads). The channel habitat is uniform and consists of unconsolidated sediment (sands and silts) that are typically subject to periodic transport. Depending on the local physical regimen in the channel, the sediment may be seasonally stable or unstable, which can result in heterogeneous infaunal communities based on sediment stability. With the exception of Willamette Cove, Swan Island Lagoon, and near-shore slips, shallow-water habitats are largely limited to the narrow strip

between the shoreline and the navigation channel (for detailed description of the Study Area including figures, see Sections 4.0 to 7.0 in the Comprehensive Round 2 Report). The infaunal communities in the shallow-water habitats are controlled by physical characteristics such as slope, grain size, and the magnitude of disturbance events that may occur, with more well-developed communities expected in low-gradient, sheltered areas.

Benthic invertebrate habitats can generally be divided into soft and hard substrates, with soft substrates supporting an infaunal and epifaunal community and hard substrates supporting an epifaunal community. A total of 117 invertebrate taxa distributed over 8 phyla were collected in the Study Area during Rounds 1 and 2 (Table 3-1). The structure and function of the infaunal and epifaunal communities are presented in Sections 3.1.1.1 and 3.1.1.2, respectively.

3.1.1.1 Infaunal Community Structure and Function

The infaunal invertebrate community within the Study Area is dominated by oligochaetes (segmented worms), chironomids (midge larvae), *Corophium* sp. (amphipods), and clams (Integral et al. 2004b). Other common invertebrates include nematodes (round worms) and polychaetes (segmented worms).

Most organisms collected in the Study Area were consistent with the type of species expected in a deep river. According to the River Continuum Concept, invertebrate communities in deep rivers are typically dominated by the feeding group known as collectors (Vannote et al. 1980). Collectors are composed of both gatherers, organisms that forage for organic matter in the sediments, and filterers, organisms that filter organic matter out of the water column (Cummins and Klug 1979). Oligochaetes, which are gatherers, feed on bacteria, diatoms, detritus, and other microorganisms by ingesting large quantities of sediment and extracting organic material. Some species live from 1 to 3 cm below the sediment surface, while others live in tubes attached to filamentous algae, submerged plants, and terrestrial debris (Brusca and Brusca 2003). Chironomids, the only infaunal dipterans (true flies), have a juvenile aquatic larval stage and become terrestrial after their metamorphosis into adult form. The larvae can be collectors/gatherers, herbivores, detritus shredders, or engulfing predators. Some species are sprawlers and clingers, attaching themselves to or lying flat against rocks and other substrate; whereas other species are infaunal burrowers, gathering organic matter in and on the sediments.

Amphipods are scavenging omnivores, feeding on various kinds of plant and animal material. The most common amphipod in the Study Area, *Corophium* spp., is described as both a gatherer and filterer (Percy 1999) and is found in fine sediments where they make U-shaped burrows (McCabe et al. 1997). The introduced Asiatic clam (*Corbicula* sp.), the most abundant bivalve in the Study Area, feed on zooplankton, phytoplankton, and organic detritus. They can occur in water up to 30 m deep but are predominantly found in water depths from 0 to 2 m. They prefer stable sand and gravel substrates and are not generally found in areas of high turbidity or

clay or rock substrate (Pennak 1978). Nematodes are free-living and parasitic roundworms; and in natural freshwater habitats, most specimens are confined to the uppermost 5 cm of the substrate (Pennak 1978). Many infaunal nematodes are direct deposit feeders and others are detritivores or microscavengers (Brusca and Brusca 2003).

Infaunal community samples were collected by LWG in the fall of 2002 from 22 locations within the Study Area to provide general qualitative information (assessment of the infaunal community in upstream or reference areas was not included in the sampling effort). The samples were collected with a van Veen (0.1-m^2) grab sampler, and the infauna were analyzed in 21 of the 22 samples. Tar in one sample made identification of the invertebrates impossible. At all locations, the infaunal community was dominated by oligochaete worms, chironomid larvae, clams, and crustaceans. The total number of taxa ranged from 6 to 21 per 0.1 m^2, and the total abundance ranged from 7 to 590 per 0.1 m^2. Chironomids were found in densities ranging from 1 to 326 per 0.1 m^2 at 18 of the 21 sampling locations (they were absent in 3 samples), usually with 2 to 5 taxa at each location. Oligochaetes had a similar distribution with a total of 12 taxa found in the Study Area, usually with 3 to 6 taxa per location. The oligochaete *Limnodrilus hoffmeisteri* was the most common taxon present at 19 of the 21 locations, with abundances ranging from 17 to 316 per 0.1 m^2. Three groups of crustaceans, amphipods, isopods, and ostracods, were found during the survey, usually in low abundances (fewer than five organisms per location) with only *Corophium* sp., a small amphipod, reported in higher densities (10 to 148 per 0.1 m^2). Clams were represented by two taxa, *Corbicula* sp. and *Pisidium* sp., at 19 of the 21 locations, with densities ranged from 1 to 191 per 0.1 m^2. The infaunal community structure in the Round 1 survey was influenced by distance from the Columbia River and percent total organic carbon (%TOC). The community structure became less diverse with distance from the Columbia River and with lower %TOC. As predicted by the River Continuum Concept, the composition of functional groups at the 21 sampling locations was dominated by filterers and gatherers and shifted towards gatherers with distance from the Columbia River.

Two benthic invertebrate sampling events were conducted by LWG in the Study Area in 2005 (Windward and Integral 2005a, b). A reconnaissance survey was conducted at 13 locations in the Study Area to assess the possibility of collecting sufficient benthic invertebrates for tissue chemistry analyses. Based on the reconnaissance survey, the second field effort collected clams (*Corbicula* sp.) at 33 locations in the Study Area. The benthic invertebrates observed in the two field efforts were similar to the organisms collected in 2002 and included chironomids, oligochaetes, clams, flatworms, and dragonfly larvae. *Corbicula* sp. was the most common larger benthic invertebrate and was collected at all 33 locations in the Study Area. Two other larger mollusks, tentatively identified in the field as western pearlshell mussel (*Margaritifera falcata*) and winged floater (*Anodonta nuttalliana*), were collected at 17 and 2 locations, respectively. Gastropod snails (Pleuroceridae) were collected in

relatively larger numbers at the confluence of Multnomah Channel and the main stem of the river.

The infaunal community structure throughout the Study Area was surveyed by LWG in 2002 using a sediment profile camera (SEA 2002). Designations of infaunal successional stages were made for each analyzed image by observing the biological features present in the image and attributing the features to functional types of benthic infauna. Stage 1 infauna were typically shallow-dwelling tube builders that fed at or near the sediment-water interface. These organisms are typically the first colonizers after a disturbance of the sediment. Stage 2 infauna were identified by dense amphipod tubes and/or the presence of shallow feeding voids.[7] Stage 3 infauna are typically deep burrowers that feed within the sediment column. They are generally longer living than Stage 1 infauna and are the slowest to recolonize a disturbed area. Because the designation of succession stages was based on invertebrates living in fine-grained sediments, the benthic communities in areas of coarser sediments would not be expected to follow the three successional stages.

The infaunal community structure in the upper segment of the Study Area (RM 7 to RM 10) was characterized by the widespread presence of Stage 3 infauna, both in nearshore areas (including Swan Island Lagoon) and main channel sediments (SEA 2002). The only areas that exhibited only Stage 1 infauna were the western main channel between RM 8.4 and RM 9.3, isolated nearshore locations, the northeast corner of Swan Island Lagoon, and the main channel of the river to the western shore between RM 7.1 and RM 7.5. These areas coincided with a region of high deposition, suggesting that the deposition occurred at a rate that exceeded the ability of any longer-living Stage 3 infauna to survive. The infaunal community structure in the middle segment of the Study Area (RM 5 to RM 7) was dominated by Stage 1 infauna (SEA 2002). The presence of Stage 1 infauna in this segment of the river appeared to be related to physical disturbance from high current flow. The infaunal community structure in the lower segment of the Study Area (RM 2 to RM 5) was dominated by Stage 3 infauna, either by themselves or in associated with Stage 1 infauna (SEA 2002). Stage 3 infauna were observed in both main channel and nearshore sediments. Areas that exhibited Stage 1 infauna exclusively were the slips, the upstream portion of this segment, and at the confluence of Multnomah Channel and the Study Area. The presence of Stage 3 infauna mirrored the distribution of fine-grained sediment (for distribution of fine-grained sediments see figures in Section 4.0 of the Comprehensive Round 2 Report). Upstream, in the middle segment of the Study Area, the sediments were coarse, indicative of high flows. The fines that were carried through the middle segment appeared to be deposited in the lower reaches of the Study Area, resulting in a fine-grained region that offered suitable habitat for the colonization of Stage 3 infauna.

[7] Pockets of sediments that have been excavated by benthic infauna through deposit feeding.

3.1.1.2 Epifaunal Community Structure and Function

The epifaunal community of the Study Area is dominated by ephemeropterans (mayflies), trichopterans (caddis flies), chironomid larvae (midges or true flies), crustaceans (both amphipods and crayfish), oligochaetes (segmented worms), platyhelminthes (flatworms), and mollusks.

During the summer of 2002, LWG collected crayfish for chemical analysis at 23 locations throughout the Study Area (SEA et al. 2003). The crayfish were not identified to species; however, only two species, the western freshwater crayfish (*Pacifastacus leniusculus*) and the Siberian prawn (*Exopalaemon modestus*), exist in the Study Area. Crayfish are omnivores with a diet composed mainly of aquatic vegetation; but they will eat fish, aquatic insects, and detritus when aquatic vegetation is scarce (Pennak 1978). Adult crayfish remain in burrows, under stones or debris or half-buried in substrate, during the day and become more active between dusk and dawn; whereas juveniles can be active during the day.

During the summer in both 2002 and 2005, LWG conducted surveys of the epifaunal communities present in the Study Area by deploying multiplate samplers made of an artificial substrate in the water column. In 2002, multiplate samplers were deployed at 10 locations in the Study Area and at 2 locations between RM 9 and RM 13, which at that time were regarded as upstream reference areas. In 2005, epifaunal invertebrates were collected for chemistry analysis at 10 different locations in the Study Area. After 6 weeks, the multiplate samplers were retrieved, and the organisms that had colonized the substrate were identified. Chironomid larvae, oligochaetes, and *Corophium* sp. dominated the epifaunal community collected on the multiplate samplers. Other common epifaunal invertebrates were sponges and bryozoa.

3.1.2 Selected Receptors of Concern

The following subsections present the rationale for the selection of the benthic receptors of concern for this ERA.

3.1.2.1 Benthic Community

Benthic invertebrates are generally evaluated as a community in ERAs. Benthic invertebrates in the Study Area integrate multiple exposure pathways, including direct contact with sediment, and therefore have a high exposure potential. They also are present in the ecosystem year-round and represent site-specific exposures. Benthic invertebrates are an important food source for other invertebrates, fish, birds, and mammals, and provide essential nutrient cycling to the Study Area ecosystem. Infaunal and epifaunal invertebrates often comprise a significant portion of the heterotrophic biomass in a river system (Jahn and Anderson 1986) and thus serve as a principal food resource for higher trophic-level consumers (i.e., fish and wildlife). Benthic invertebrates also control energy flow by acting as principal processors of organic matter (Merritt et al. 1984). In large, deep rivers, such as the Study Area, particles suspended in the water column are a significant source of organic matter for the ecosystem. In addition, benthic organisms have been shown to be susceptible to

sediment-associated chemicals, and data are available to assess their exposure and predict potential effects.

3.1.2.2 Shellfish

Potential adverse effects to shellfish (bivalves) are assessed separately from those to the benthic community as directed by EPA in the Programmatic Work Plan (Integral et al. 2004b). Bivalves are important filter feeders, capturing energy from the water column, and serve as potential food resources for higher trophic-level consumers. In addition, they are sessile (not free-moving) and thus provide information on site-specific sediment conditions.

3.1.2.3 Crayfish

Potential adverse effects to crayfish are assessed separately from those to the benthic community as directed by EPA in the Programmatic Work Plan (Integral et al. 2004b). Crayfish are important prey species. They have small home ranges and thus may be susceptible to localized concentrations of chemicals. Crayfish are also highly exposed because of their direct contact with sediment, and they have a relatively long life span (up to 8 years) compared to other invertebrates (Hobbs 2001).

3.1.3 Exposure Pathways

Potential chemical exposure pathways for the benthic invertebrate receptors in the Study Area were presented in Appendix B of the Programmatic Work Plan (Integral et al. 2004b). The exposure pathways were categorized as complete and major, complete and minor, complete and uncertain, and incomplete (Figure 3-1). The complete and major pathways are quantitatively assessed, and the complete and minor pathways are discussed qualitatively to a level of certainty dependent on available studies. The complete and uncertain pathways and incomplete pathways associated with TZW are assessed quantatively at nine locations in the Study Area. As shown in Figure 3-1, benthic invertebrate receptors may be exposed to chemicals in water (including surface water and TZW) or in sediment either directly through contact with or ingestion of sediment or water or indirectly through the food chain. Complete and major pathways for benthic invertebrates (infauna and epifauna) are dermal contact with sediment and surface water and ingestion of sediment and biota. Ingestion of surface water is regarded a complete and minor pathway. Complete and major pathways for crayfish are ingestion of sediment and biota and dermal contact with surface water. Ingestion of surface water and dermal contact with sediment are regarded as complete and minor pathways. Ingestion of and dermal contact with TZW are regarded as incomplete pathways for epifauna and crayfish and complete and uncertain for infauna.

3.1.4 Lines of Evidence Approach and Methods

The measures of exposure and effects and LOEs for assessing risks to benthic invertebrates were presented in Appendix B of the Programmatic Work Plan (Integral et al. 2004b). These LOEs were further refined based on discussions between LWG,

EPA, and EPA's partners (EPA 2004a, 2005a). Table 3-2 presents current agreed-upon LOEs that were used in the risk assessment of the benthic community, shellfish, and crayfish. One LOE – literature-based SQVs (both consensus-based and empirical) – was assigned zero weight because it failed to predict site-specific measurements of benthic toxicity (see Attachment G2). The weighting scheme for the other LOEs depended on the purpose of the assessment/task, of which there were four:

1. Screening COIs to identify Round 2 COPCs
2. Further screening of Round 2 COPCs to identify iCOCs
3. Identifying potential data needs for the BERA
4. Providing information that would be used to identify iAOPCs

Round 2 COPCs were identified in the SLERA. The other three tasks were done to better prepare for the BERA and other parts of the RI/FS that may use risk assessment results.

The weight-of-evidence approach for Round 2 COPC identification was as follows:

- Round 2 COPCs were identified for each LOE (except SQVs, which were assigned a weight of zero).
- The Round 2 COPC lists were integrated across LOEs to get the overall list of Round 2 COPCs for benthic invertebrates.

This is a conservative approach in that any chemical that screened in based on any one LOE was carried forward as a Round 2 COPC. The identification of iCOC used the same weight-of-evidence approach as Round 2 COPC identification, differing only in level of conservatism. Round 2 COPC identification used an appropriate level of conservatism for a SLERA. The iCOC identification used less-conservative assumptions and incorporated a weighted approach across LOEs in an effort to narrow the list of chemicals and identify the potential areas associated with risks to benthic invertebrates in the Study Area. This was important for identifying data needs and for developing iAOPCs that anticipate where the AOPCs (to be developed in the BERA) will be.

The weight-of-evidence approaches for assessing data needs and identifying iAOPCs differed from the weight-of evidence approach for Round 2 COPC and iCOC identification because of the different objectives of these tasks.

For these tasks the LOEs were used sequentially for data needs assessment and iAOPC identification based on a hierarchy of site-specificity:

- Toxicity testing uses site-specific effects data.

- Predictive modeling uses site-specific exposure and effect data to predict toxicity where empirical toxicity test data are unavailable.
- The other LOEs compare site-specific tissue, water and sediment concentration data with generic screening values. These LOEs are more directly comparable to the fish and wildlife LOEs.

The data needs assessment focused on areas a) without toxicity test data and b) where the toxicity classification based on predictive modeling was "indeterminate." These represent the areas of greatest uncertainty. The other LOEs for these locations and all LOEs from nearby locations – including nearby toxicity test data and predictive modeling as well as fish and wildlife LOEs – were considered in the data needs assessment.

iAOPC identification focused on areas that toxicity test data or predictive modeling identified as "toxic." The other LOEs were evaluated for areas classified as "indeterminate" based on predictive modeling, but for purposes of the Round 2 report no rigorous weight-of-evidence framework was used to identify iAOPCs based on the other LOEs.

The remainder of this section is organized by LOE. Toxicity testing and predictive modeling LOEs are presented in Section 3.2. Tissue-residue-based LOEs are presented in Section 3.3. Water concentration-based LOEs are presented in Sections 3.4 and 3.5.

3.1.5 Screening Summary for Round 2 COPC Identification

Two measures of exposure and effects, tissue data (empirical and predicted) compared to tissue-residue TRVs and water data compared to AWQC or TRVs, included an initial screening process for identification of Round 2 COPCs. This section presents a brief summary of the screening process.

The first step in the initial screening process was identification of COIs. For the tissue evaluation any chemical detected in the tissue samples were identified as COIs. Similar for the water evaluation, any chemical detected in either near-bottom surface water samples or TZW samples were identified as COIs. The maximum concentrations of chemicals identified as COIs in the tissue samples were then compared to the respective aquatic TRV. If the maximum concentration exceeded the TRV the chemical was identified as a Round 2 COPC. For the predicted tissue evaluation, COIs were identified as COPCs if the predicted 95th percentile tissue concentration exceeded the TRV. The water evaluation followed the same approach as for empirical tissue, the maximum near-bottom surface water concentration or TZW concentration was compared to the chronic Eco SL. If the maximum concentration exceeded the chronic Eco SL the chemical was identified as a Round 2 COPC.

The initial screening process is presented in Attachment G2, and the following Round 2 COPCs were identified based on each LOE:

- Seventeen Round 2 COPCs (arsenic, cadmium, copper, zinc, TBT, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, pyrene, total PAHs, BEHP, dibutyl phthalate, total PCBs, 4,4′-DDD, and total DDTs) were identified based on both field-collected and laboratory-exposed tissue samples.
- Eleven chemicals (antimony, arsenic, cadmium, copper, zinc, TBT, benzo(a)pyrene, pyrene, total PAHs, beta-HCH, and endrin) were identified as Round 2 COPCs based on the predictive tissue approach.
- Ten Round 2 COPCs (zinc, benzo(a)anthracene, benzo(a)pyrene, 4-chloro-3-methylphenol, total PCBs, 2,4′-DDD, 2,4′-DDT 4,4′-DDD, 4,4′-DDT, and total DDTs) were identified based on near-bottom surface water samples.
- Fifty-three Round 2 COPCs (including 8 metals, dalapon, Silvex™, 16 PAHs, 2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDE, 4,4′-DDT, total DDTs, 1,2-dichlorobenzene, 1,4-dichlorobenzene, dibenzofuran, 16 VOCs, perchlorate, and cyanide) were identified based on the TZW samples, and two Round 2 COPCs (acenaphthene and acetone) were idenitified based on the predicted TZW concentrations using the equilibrium partitioning (EqP) approach.

3.2 SEDIMENT TOXICITY TESTING AND MODELING

The exposure of and effects on the benthic community related to sediment were assessed by measuring toxicity using two toxicity tests and by predicting effects using two models. The toxicity testing approach is presented in Section 3.2.1 and the two models are presented in Section 3.2.2.

3.2.1 Sediment Toxicity Testing

Adverse effects to *Chironomus tentans* and *Hyalella azteca* caused by exposure to sediments collected at 227 locations within the Study Area and 6 upstream locations were assessed using four endpoints: survival and growth in both the *Chironomus* and *Hyalella* tests. Biological effects were defined to correspond conceptually to “minor effects level” (Level 2), and “moderate effects level” (Level 3). The two levels were based on an approach suggested by EPA, NOAA, and the Oregon Department of Environmental Quality (ODEQ) for this project and were defined as 80 and 70% of the response observed in the control sediment (Windward et al. 2006) (Table 3-3). A combined biological effect based on the four endpoints was assigned the maximum adverse effects level of the four endpoints.

The biological effects levels are based on statistically significant differences from the negative control (based on a one-tailed, parametric or non-parametric *t*-test, with α = 0.05) in addition to minimum difference thresholds (Table 3-3). The decision to use the negative control in the comparison was made in cooperation with EPA and its partners because of the greater reliability[8] observed using this approach in Washington and Oregon, the fact that standardized freshwater reference sites are not yet available in the region, and because the results are more conservative (Ecology 2002). At either of these effects levels, a toxicity test endpoint response is considered a hit if the difference in response is greater than the defined threshold and is statistically different from control; a no-hit station has a difference less than the threshold or is not statistically different from control. If the observed difference exceeds the threshold but is not statistically significant, the test must have had a minimum detectable difference (MDD) equal to or less than the threshold. Indeterminate stations were defined as those that had actual differences that exceeded the threshold, non-significant statistical results, and an MDD greater than the threshold. MDDs were determined for each sample comparison using *post hoc* power analysis with 80% power, one-tailed α = 0.05, and the sample variances. This process ensured that large-magnitude differences were not designated as no-hits based on lack of statistical significance because of low power.

3.2.2 Benthic Invertebrate Models

To assess areas within the Study Area where sediment toxicity testing had not been performed, two models were chosen to develop a predictive relationship between sediment chemistry and toxicity response. The two models, the FPM and the LRM, develop the predictive relationship from different perspectives, although the goals of the models are similar: to develop a predictive relationship based on empirical data (i.e., sediment chemistry and toxicity test data) and to identify the principal chemical(s) that appear to represent the relationship between sediment quality and toxicity. The FPM focuses on identifying the chemicals that are likely associated with observed toxicity and establishing SQVs for those chemicals based on minimizing errors (e.g., false positives and false negatives) and optimizing predictive reliability. The LRM focuses on developing statistical models (using logistic regression) that describe the relationship between the probability of toxicity and chemical concentrations for each chemical. In addition to developing predictive statistical models for individual chemicals, the LRM can also be used to combine multiple chemicals into a single logistic curve that provides a probability of toxicity for the chemical suite being considered.

The two models and the toxicity predictions for the Study Area sediments were presented in detail in the *Portland Harbor Superfund Site Ecological Risk Assessment: Interpretive Report: Estimating Risks to Benthic Organisms Using Predictive Models Based on Sediment Toxicity Tests* (Windward et al. 2006), hereafter referred to the Benthic Interpretive Report. Through discussion with EPA

[8] Reliability: correct predictions/total number of predictions.

and its partners, the two models were revised (EPA 2006c; LWG 2006b). The LRM was revised by NOAA on behalf of EPA and its partners and is included in this report as it was provided to LWG. The results based on the two revised models are presented in the following sections.

3.2.2.1 Data Used in the Models

The predictive models were developed based on the 233 samples with co-located sediment chemistry and toxicity test data. The LRM included additional freshwater data from a national database. The toxicity test data included the 10-day sediment toxicity test with *Chironomus tentans* and the 28-day sediment toxicity test with *Hyalella azteca*. Using the models, the risk to benthic communities was predicted for 1,031 surface sediment samples in the ERA database with chemistry data only. The ERA surface sediment chemistry data used in the models are summarized in Section 2.0.

For the FPM development, only detected values for chemicals were used because undetected chemistry values do not provide useful information for the development of a predictive relationship between sediment chemistry and benthic invertebrate toxicity. As requested by EPA and its partners, chemical data qualified with N, NJ, and NJT[9] qualifiers (specifically, Round 2 polychlorinated biphenyl [PCB] Aroclor and DDT data) were included in the model (EPA 2006c). Chemicals with fewer than 30 detected values were excluded from the modeling effort because this was the minimum threshold for a usable distribution for the development of SQVs based on analyses of other datasets from Oregon and Washington (Avocet 2003). The summation rules presented in Attachment G1 were used for the calculation of totals (e.g., use of one-half DL for non-detects).

For the LRM development performed by NOAA, all N-qualified data were included, and only detected concentrations were used.[10] For calculation of the sums for polycyclic aromatic hydrocarbons (PAHs) and PCBs used in the development of the models, NOAA's summation rules were used[11]. The sum of the PAHs excluded several analytes that are typically included in these sums (i.e., benzo(b, j, k) fluoranthenes, benzo(g,h,i) perylene, and indeno(1,2,3-cd)pyrene were excluded from total high-molecular-weight PAHs [HPAHs] and acenaphthylene was excluded from total low-molecular-weight [LPAHs]). For the application of the LRM to predict toxicity in the chemistry only samples, NOAA's total PAH values were used, while

[9] N-qualifier signifies: presumptive evidence of analyte; for metals the matrix spike sample recovery was not within control limits and for organics the identification was tentative; the analyte exhibits low spectral match parameters but was present. J-qualifier signifies: estimated value. T-qualifier signifies: value is an average or selected result (following standard project rules).
[10] For three endpoints (antimony, gamma-HCH, and total DDTs), results from preliminary models that included BDL (below detection limit) data at half the detection limit were used. A comparison of the models including and excluding the BDL data showed virtually identical results for these endpoints.
[11] Non-detects were treated as 0 unless all constituents were below detection, in which case the highest reported detection limit was used for the sum.

the LWG values were used for total PCBs and DDTs.[12] In the development of the combined (i.e., multiple chemical) model, predictions from non-detects were based on one-half DL but were ignored if the predicted probability value was greater than 0.2. Chemicals with fewer than 100 detected values were excluded from this modeling effort.

Analytes that are derived quantities (e.g., dioxin toxic equivalents [TEQs]) and chemicals that are identified as highly correlated with each other (e.g., PAHs) were represented as totals in both models. For the LRM, individual PAH, LPAH, and HPAH; total PAH; individual DDT isomers; total DDT (sum of six isomers); and total PCBs were included in model development. Chlordanes were included as individual isomers. For the FPM, totals were calculated for PCB Aroclors, total PAHs, LPAHs, HPAHs, sum DDD, sum DDE, sum DDT, total DDTs, total chlordane, and total endosulfan.[13]

The two modeling approaches had slightly different rules for including individual chemical endpoints. Within the FPM, the final SQVs are a function of the joint distribution of all chemicals present in the Study Area. The presence of non-toxic, naturally occurring crustal elements such as aluminum and selenium can confound the development of meaningful SQVs for the remainder of the analytes. Consequently, aluminum and selenium were excluded from the FPM. In the LRM approach, individual regression models were developed for each analyte independent of the concentrations of other analytes. In the final multi-chemical model,[14] the contribution of non-toxic elements to the overall predictions of toxicity could be evaluated. Consequently, including highly correlated, non-toxic analytes in the LRM did not affect the outcome, so selenium and aluminum were initially included.

Certain conventional analytes, such as specific gravity and total solids, were screened out of both models because they are not considered contaminants. However, other conventional analytes, including percent fines, bulk sediment ammonia, and sulfides, were retained in the two models because of their apparently strong correlation with toxicity in some biological endpoints. The analytes screened out as a result of the above factors are presented in Table 3-4.

3.2.2.2 Floating Percentile Model

The basic concept behind the FPM is for the user to select an optimal percentile of the dataset that provides a low target false negative rate (i.e., rate associated with erroneously concluding the sediments are not toxic) and then adjust the candidate criterion for each individual chemical concentrations upward until false positive rates

[12] A comparison of the two sets of totals for PCBs and DDTs indicated that the values were nearly identical.
[13] The models were used to identify the appropriate sums to use, based on model performance. Because the models use two very different methods (one models each chemical individually and then combines them; the other uses all chemicals simultaneously), different summing approaches may give better results for each model.
[14] The method of combining the LRM results for individual chemicals into a multi-chemical model is described in Step 9 of Section 5.3.1 of the Benthic Interpretive Report (Windward et al. 2006).

(i.e., rate associated with erroneously concluding that sediments are toxic) are decreased to their lowest possible level while retaining the same low false negative rate. Once the candidate criterion for each chemical has been individually adjusted upward to the lower end of the chemical's toxicity range, the false positive rate will have been significantly reduced while a low false negative rate is retained. This adjustment process is unlike most other existing SQV sets, which require the SQVs for all chemicals to be based on the same percentile of the hit or no-hit distribution. After the adjustment process most chemicals should be at or near a level associated with toxicity in the dataset, rather than at a level arbitrarily assigned by a fixed percentile. In this manner, optimized site-specific SQVs can be developed for a number of different target false negative rates, allowing the trade-offs between false negatives and false positives to be evaluated relative to risk management and a final set of SQVs to be selected. The COIs included in the FPM are presented in Table 3-5. The FPM is presented in detail, including a step-by-step process of how to use the model, in the Benthic Interpretive Report (Windward et al. 2006).

The toxicity predictions were based on Effects Levels 2 and 3 for the following three endpoints: *Chironomus* growth, *Chironomus* mortality, and *Hyalella* mortality. The *Hyalella* growth endpoint was not included in the model because this endpoint largely does not respond to Study Area COPCs (Windward et al. 2006). Its reliability is poor at all effects levels and greatly reduces the reliability of the pooled endpoint and the overall model. The *Hyalella* growth endpoint seems to be responding primarily to percent fines and ammonia.[15]

The reliability of the predictions based on Effects Levels 2 and 3 and the three endpoints (*Chironomus* growth, *Chironomus* mortality, and *Hyalella* mortality) was good with false negative rates ranging from 11 to 21%, false positive rates from 7 to 24%, and overall reliability ranging from 76 to 93%. The highest false negative and positive rates and the lowest overall reliability were observed at Effects Level 2 for *Chironomus* mortality.

3.2.2.3 Logistic Regression Model

In contrast to developing SQVs using a threshold approach such as FPM, the LRM approach estimates a functional curve relationship between concentrations of a chemical and the proportion of toxicity observed in the sediment toxicity tests. This enables users to select the specific probability of effects based on a risk management that corresponds to their specific assessment objective. The LRM is presented in detail including a step-by-step process of how to use the model in the Benthic Interpretive Report (Windward et al. 2006). In a parallel effort with the Benthic Interpretive Report, NOAA developed alternative LRMs using a larger freshwater database for the *Hyalella* 28-day growth and survival endpoint and calibrated[16] these

[15] The issue of ammonia toxicity is discussed in the Benthic Interpretive Report (Windward et al. 2006).
[16] Calibration was completed following the approach described in Step 9 of Section 5.3.1 of the Benthic Interpretive Report (Windward et al. 2006).

models to the pooled (survival and growth, combined) Level 2 Effects for all *Hyalella* and *Chironomus* endpoints in the Portland Harbor data (Field 2006).

The non-site-specific freshwater effect database used by NOAA contained 634 synoptic samples with both chemistry and *Hyalella* 28-day survival and growth responses (233 samples from Portland Harbor, 100 from Calcasieu Estuary, and the remainder from the Great Lakes and other locations throughout North America) (Field 2006). The process involved fitting a suite of individual LRMs to each of the COIs in Table 3-6.

The suite of models included different approaches to initial screening and exclusion of potential outliers and different degrees of response (lowest percent of control-adjusted response in either growth or survival endpoints). For each chemical, individual models within the suite were scored based on predictive performance for the Portland Harbor dataset. The best model for each chemical was retained; this process did not restrict the set of final chemical models to all that had the same model rule (i.e., using the same screening process or exclusion of outliers for every chemical). The set of selected individual models were combined into a multiple chemical model using the concentration with the maximum probability of effect for each sample from any individual model. Individual models with high false positive rates were eliminated from the combined model. Because models for total PAH provided comparable performance to the individual PAH models, they were used in the combined model. Models for percent fines and aluminum were excluded from the combined models. The calibration of the multi-chemical model was based on Portland Harbor Level 2 Effects (all *Hyalella* and *Chironomus* endpoints pooled).[17]

3.3 TISSUE-RESIDUE ASSESSMENT

The critical tissue-residue assessment was one LOE for evaluating risks to selected benthic invertebrate receptors. Exposure of and effect to the benthic community was assessed by comparing tissue-residue concentrations to TRVs available in the literature. This LOE incorporated all of the exposure pathways for benthic invertebrates. The tissue-residue data were derived using field-collected and laboratory-exposed organisms, which are presented as two LOEs in Table 3-2. Field-collected tissue samples included clams (*Corbicula* sp.), crayfish, and invertebrates[18] collected with multiplate samplers; laboratory-exposed tissue samples were obtained from clams (*Corbicula fluminea*) and worms (*Lumbriculus variegatus*). In addition, predicted tissue concentrations were compared to TRVs. Use of the two tissue types and the predicted tissue-residue assessment in the risk assessment is summarized in Table 3-2.

[17] The general method for calibration is described in Step 9 of Section 5.3.1 of the Benthic Interpretive Report (Windward et al. 2006).
[18] Epibenthic invertebrates and zooplankton.

The initial screening process for the tissue-residue assessment is presented in Attachment G2. The screening process included selection of COIs and identification of Round 2 COPCs by comparing the maximum concentrations measured in each tissue type to aquatic TRVs. Seventeen Round 2 COPCs were identified based on both field-collected and laboratory-exposed tissue samples (Table 3-7). Eleven chemicals, antimony, arsenic, cadmium, copper, zinc, TBT, benzo(a)pyrene, pyrene, total PAHs, beta-HCH, and endrin, were identified as Round 2 COPCs based on the predictive tissue approach (Table 3-7). The initial screening process for the predicted tissue assessment is also presented in Attachment G2. The screening process included the selection of COIs and identification of Round 2 COPCs by comparing the predicted 95th percentile tissue concentrations to aquatic TRVs.

3.3.1 Round 2 COPC Exposure Assessment

This section presents the exposure assessment using empirical data collected in the field and laboratory for the following benthic invertebrate tissue types: field-collected clams (Section 3.3.1.1), field-collected crayfish (Section 3.3.1.2), invertebrates collected with multiplate samplers (Section 3.3.1.3), laboratory-exposed clams (Section 3.2.1.4) and laboratory-exposed worms (Section 3.3.1.5). In addition, the exposure assessment for the predicted tissue approach is presented in Section 3.3.1.6. Figure 3-2 presents all of the locations where tissue samples were collected or where sediments were collected for the laboratory exposure of clams and worms.

The upper bound estimate (e.g., an upper confidence limit [UCL]) was calculated for all tissue Round 2COPCs using the ProUCL software. ProUCL software was developed by EPA to compute an appropriate UCL of an unknown population mean. ProUCL tests for normality, lognormality, and a gamma distribution of the dataset and computes a conservative and stable UCL of the unknown population mean (EPA 2004c). Before the data were run through ProUCL to determine UCLs, non-detected data were adjusted to one-half the detection limit to represent the concentration value. The ProUCL-recommended UCL for each Round 2 COPC was used in the risk characterization. In cases where insufficient data were available, the maximum concentration was used. Attachment G1 presents the summary statistics (i.e., minimum, maximum, and mean COPC concentrations), distribution types, and ProUCL-recommended UCLs.

3.3.1.1 Field-Collected Clam Tissue Data

A summary of the nine chemicals identified as Round 2 COPCs based on field-collected clams is presented in Table 3-8. Three metals (cadmium, copper, and zinc) were detected in all clam tissue samples,[19] with several concentrations near the upper ranges presented in Table 3-8. Tributyltin (TBT) was detected in 78% of the tissue samples. The detected concentrations in tissue ranged from 2.5 to 33 µg/kg ww,

[19] The following five issue samples out of the 33 tissue samples were not analyzed for metals because of limited biomass: BTFC011, BTFC018, BTFC029, BTFC032, and BTFC033

except in two tissue samples collected in the navigation channel at the mouth of the International Slip (BT005) and adjacent to Portland Shipyard (BT023), which had TBT concentrations of 63 and 530 µg/kg ww, respectively.

Total PAHs were detected in all field-collected clam tissue samples; concentrations ranged from 34.8 to 720 µg/kg ww, with the exception of four tissue samples collected downstream of the Arco (BT012), in the US Moorings embayment (BT014), adjacent to Gasco (BT015), and downstream of Arkema (BT017). These four tissue samples had concentrations ranging from 1,010 to 4,980 µg/kg ww. All tissue concentrations of dibutyl phthalate were qualified as a non-detect (U) and ranged from 8 to 250 µg/kg ww, except in one tissue sample collected at the mouth of Swan Island Lagoon (BT022), which had a detected concentration of 1,300 µg/kg ww.

Total PCBs were detected in all field-collected clam tissue samples; concentrations ranged from 50.1 to 513 µg/kg ww, with the exception of one tissue sample collected in Willamette Cove (BT016), which had a total PCB concentration of 2,660 µg/kg ww. The pesticide 4,4′-DDD was detected in all tissue samples. Concentrations ranged from 1.76 to 30 µg/kg ww, with the exception of two samples collected adjacent to or downstream of Arkema (BT017 and BT018) during Round 2 and in one sample collected near Arkema (07R006) during Round 1. These three tissue samples had concentrations that ranged from 88.8 to 160 µg/kg ww. Total DDTs were detected in all tissue samples. Concentrations ranged from 7.82 to 181 µg/kg ww, with the exception of one sample collected downstream of Arkema (BT017) during Round 2 and one sample collected near Arkema (07R006) during Round 1. Concentrations in these two tissue samples were 304 and 463 µg/kg ww.

3.3.1.2 Crayfish Tissue Data

Copper was the only Round 2 COPC identified for field-collected crayfish (Table 3-9) and was detected in all tissue samples, with concentrations ranging from 10.4 to 17.6 mg/kg ww.

3.3.1.3 Multiplate Tissue Data

Copper was the only Round 2 COPC identified based on invertebrates collected in the field using multiplate samplers (Table 3-10). Invertebrate tissue samples were collected at 10 locations in the Study Area. However, because of limited tissue mass, samples from five locations were composited into two samples. Two out of the seven samples were analyzed for metals. Copper was detected in both samples with concentrations of 3.01 and 6 mg/kg ww (MIT009 and MIT001).

3.3.1.4 Laboratory-Exposed Clam Tissue Data

A summary of the six chemicals identified as Round 2 COPCs based on laboratory-exposed clams is presented in Table 3-11. Copper was detected in all tissue samples with several concentrations near the upper range presented in Table 3-11. TBT was detected in 26% of the tissue samples. The tissue concentrations ranged from 1.1 to

34 µg/kg ww, except in one tissue sample exposed to sediment collected at Portland Shipyard (BT023) with a concentration of 680 µg/kg ww.

Total PAHs were detected in all laboratory-exposed clam tissue samples. The tissue concentrations ranged from 19.2 to 845 µg/kg ww, except for one tissue sample exposed to sediment collected downstream of Arco (BT012) with a concentration of 1,320 µg/kg ww. One chemical, bis(2-ethylhexyl) phthalate (BEHP), was identified as a Round 2 COPC based only on laboratory-exposed clams. The chemical was detected in 77% of the tissue samples with concentrations ranged from 53 to 160 µg/kg ww, except for one tissue sample exposed to sediment collected adjacent to Gunderson (BT028) with a concentration of 8,600 µg/kg ww.

The pesticide, 4,4′-DDD, was detected in all laboratory-exposed clam tissue samples. The tissue concentrations ranged from 0.375 to 84.5 µg/kg ww, except for one tissue sample exposed to sediment collected adjacent to Arkema (BT018) with a concentration of 702 µg/kg ww. Total DDTs were detected in all tissue samples. The tissue concentrations ranged from 1.1 to 116 µg/kg ww, except for one tissue sample exposed to sediment collected adjacent to Arkema (BT018) with a concentration of 1,040 µg/kg ww.

3.3.1.5 Laboratory-Exposed Worm Tissue Data

A summary of the 16 chemicals identified as Round 2 COPCs based on laboratory-exposed worms is presented in Table 3-12. Four metals (arsenic, cadmium, copper, and zinc) were detected in all tissue samples with several concentrations near the upper ranges presented in Table 3-12. TBT was detected in 43% of the tissue samples. The tissue concentrations ranged from 2.1 to 49 µg/kg ww, except in one tissue sample exposed to sediments collected adjacent to Portland Shipyard (BT023), which had a concentration of 1,700 µg/kg ww.

Six individual PAHs identified as Round 2 COPCs were detected in all laboratory-exposed worm tissue samples. Most concentrations of these PAHs were below 1,000 µg/kg ww, except in a few tissue samples (1 to 5) with concentrations ranging between 1,400 to 11,000 µg/kg ww. These samples predominately consisted of worms exposed to sediments collected downstream of Arco (BT012), in US Mooring embayment (BT014), near Gasco (BT015), and downstream or adjacent to Arkema (BT017 and BT018). Total PAHs were calculated for all tissue samples with several concentrations near the upper range presented in Table 3-12. Dibutyl phthalate was detected in 11% of the tissue samples and was identified as a Round 2 COPC based on qualified data (J). All concentrations were less than or equal to 175 µg/kg ww except one estimated concentration (470J) in worms exposed to sediment collected at the mouth of Saltzman Creek (BT020).

The majority of total PCBs tissue concentrations measured in laboratory-exposed worms ranged from 44.8 to 656 µg/kg ww. Six tissue samples from the following locations had total PCBs concentrations above this range: Swan Island Lagoon (730 µg/kg ww at BT026), Balch Creek Cove (1,450 µg/kg ww at BT032),

downstream of Arkema (1,890 µg/kg ww at BT017), Oregon Steel Mill (2,970 µg/kg ww at BT002), Willamette Cove (3,910 µg/kg ww at BT016), and adjacent to Gunderson (4,310 µg/kg ww at BT028).

The pesticides 4,4′-DDD and total DDTs, were detected in all laboratory-exposed worm tissue samples. The majority of the tissue samples had concentrations of 4,4′-DDD ranging from 4.08 to 61.1 µg/kg ww except in three samples. These samples were exposed to sediments collected downstream or adjacent to Arkema (BT017 and BT018), and adjacent to Gunderson (BT028) and had tissue concentrations of 1,060, 705, and 113 µg/kg ww. Similarly, the majority of the tissue samples had total DDTs concentrations ranging from 14.5 to 284 µg/kg ww except in two samples. These samples were exposed to sediments collected downstream or adjacent to Arkema (BT017 and BT018) and had tissue concentrations of 1,490 and 1,020 µg/kg ww.

3.3.1.6 Predicted Tissue Data

Eleven chemicals were identified as Round 2 COPCs for benthic invertebrates through the predicted tissue screening process presented in Attachment G2. The Round 2 COPCs included cadmium, zinc, TBT, and total PAHs based on field-collected clams; copper, TBT, pyrene, total PAHs, beta-HCH, and endrin based on laboratory-exposed clams; and antimony, arsenic, TBT, benzo(a)pyrene, pyrene, total PAHs, beta-HCH, and endrin based on laboratory-exposed worms. The predicted site-wide tissue concentrations for these Round 2 COPC/receptor pairs based on site-wide UCL sediment concentrations are presented in Table 3-13.

3.3.2 Round 2 COPC Effects Assessment

This section presents a discussion of the TRV selection process to address invertebrate-specific toxicity (the aquatic TRVs used in the Round 2 risk characterization were based on all aquatic phyla) and the associated toxicity literature for each of the 20 Round 2 COPCs identified based on invertebrate tissue data and the predicted tissue approach. The Round 2 COPCs based on empirical data included four metals (arsenic, cadmium, copper, and zinc), TBT, six individual PAHs, total PAHs, two phthalates (BEHP and dibutyl phthalate), total PCBs, and two pesticides (4,4′-DDD and total DDTs). The Round 2 COPCs based on the predictive tissue approach included antimony, arsenic, cadmium, copper, zinc, TBT, benzo(a)pyrene, pyrene, total PAHs, beta-HCH, and endrin.

TRVs for these chemicals were selected in three different ways. For total PCBs, cadmium, total DDTs, and BEHP, a sufficient number of studies (at least 20 studies) was available in the literature and the TRV was selected as the fifth percentile of all aquatic species LOAEL data (including fish, crayfish, and clam). This approach, using species sensitivity distribution, was intended to define a tissue-residue concentration that was protective of 95% of aquatic species. An insufficient number of studies (less than 20 studies) was available in the literature for arsenic, copper, zinc, and endrin. TRVs were selected from Dyer et al. (2000), which reported the fifth

percentile LOAEL for these chemicals based on fish data. Similar to the approach above these TRVs were intended to define a tissue-residue concentration that was protective of 95% of aquatic species. For the remaining 12 Round 2 COPCs, an insufficient number of studies was available in the literature, and the chemicals were not evaluated in Dyer et al. (2000). A tissue-residue TRV was calculated as the product of EPA AWQC and a bioconcentration factor (BCF) (see Attachment G2). Because the selected TRVs were based on data which included fish and other non-benthic invertebrates a short discussion is provided for each Round 2 COPC comparing the selected TRVs to toxicological studies with benthic invertebrates.

3.3.2.1 Antimony

A literature search found no toxicological studies with decapods and mollusks that reported tissue concentrations of antimony associated with adverse effects (Windward 2005).

3.3.2.2 Arsenic

Only two studies with invertebrates reported tissue concentrations of arsenic associated with adverse effects (Lindsay and Sanders 1990; Spehar et al. 1980) (Table 3-14). No effect was observed in grass shrimp with a tissue concentration of 1.28 mg/kg ww after exposure to arsenic in the water for 28 days. Similarly, no effect was observed in freshwater snails exposed to arsenic in the water for 28 days. The reported no-observed-adverse-effect level (NOAEL) tissue concentrations ranged from 1.15 to 16 mg/kg ww. The selected arsenic TRV of 1.7 mg/kg ww, which is calculated as the fifth percentile of the lowest-observed-adverse-effect levels (LOAELs) based on fish data, is within the concentration range of NOAELs observed for invertebrate species.

3.3.2.3 Cadmium

Seven studies with crustaceans reported tissue concentrations of cadmium associated with adverse effects (Carr et al. 1985; Gillespie et al. 1977; Mirenda 1986b; Reddy et al. 1997; Rule and Alden 1996; Thorp et al. 1979; Vernberg et al. 1977) (Table 3-14). These studies reported the effects of cadmium on survival, growth, or reproduction in crayfish, grass shrimp, or mysid shrimp. LOAELs ranged from 1.29 mg/kg ww for growth of shrimp (Carr et al. 1985) to 28.4 mg/kg ww for mortality of crayfish (Mirenda 1986b). At the lowest LOAEL of 1.29 mg/kg ww, a significant reduction in growth (17%) was reported in mysid shrimp following a 33-day exposure to cadmium in water. NOAELs ranged from 0.5 mg/kg ww for reproduction of crayfish (Reddy et al. 1997) to 534 mg/kg ww for mortality of crayfish (Gillespie et al. 1977). Six studies with mollusks reported tissue concentrations of cadmium associated with adverse effects (Carlson et al. 1991; Rule and Alden 1996; Spehar et al. 1978; Tessier et al. 1996; Zaroogian 1980; Zaroogian and Morrison 1981). These studies reported the effects of cadmium on survival or reproduction in clams, snails, Eastern oysters, and eastern elliptios. LOAELs ranged from 30 mg/kg ww (Spehar et al. 1978) to 625 mg/kg ww (Carlson et al. 1991) for mortality of snails. At the lowest LOAEL of 30 mg/kg ww, 90% mortality was reported in freshwater snails exposed to cadmium

in water for 28 days (Spehar et al. 1978). NOAELs ranged from 0.46 mg/kg ww for mortality of clams (Rule and Alden 1996) to 460 mg/kg ww for mortality of snails (Carlson et al. 1991). The selected cadmium TRV of 0.09 mg/kg ww, which is calculated as the fifth percentile of all aquatic species LOAEL data in the Ecological PRE (Windward 2005), is a conservative value, lower than all NOAELs reported for benthic invertebrates with a factor ranging from 6 and to 5,900 (approximately 6 orders of magnitude).

3.3.2.4 Copper

Two studies with decapods reported tissue concentrations of copper associated with adverse effects (Evans 1980; Rule and Alden 1996) (Table 3-14). No effect on grass shrimp or crayfish mortality was observed in either study following exposure to copper. The two NOAELs were reported as 40 and 50 mg/kg ww. Four studies with bivalve species reported tissue concentrations of copper associated with adverse effects on survival (Absil et al. 1996; Kumaraguru et al. 1980; Roesijadi 1980; Rule and Alden 1996). LOAELs ranged from 9.34 mg/kg ww (Roesijadi 1980) to 201 mg/kg ww (Kumaraguru et al. 1980). At the lowest LOAEL of 9.34 mg/kg ww, (Roesijadi 1980) reported 17% mortality in marine Pacific littleneck clams exposed to copper in water for 30 days; 3% mortality was reported in control clams. The only NOAEL reported in the literature was based on the results of Rule et al.(1996). At 6.4 mg/kg ww, no significant mortalities were observed in a bioaccumulation study in which clams were exposed to copper over a period of only 14 days. The selected copper TRV of 3.1 mg/kg ww that was calculated as the fifth percentile of LOAELs based on fish data is a conservative value, lower than all NOAELs reported for benthic invertebrates by a factor ranging from 2 to 16. However, although all invertebrate studies evaluated adverse effects on survival, none included sublethal effects.

3.3.2.5 Zinc

Only one study with decapods reported tissue concentrations of zinc associated with adverse effects (Mirenda 1986a) (Table 3-14). Crayfish were exposed to zinc through water exposure for 2 weeks. Twenty-three and six percent mortality was observed in crayfish with tissue residues of 35.2 and 12.7 mg/kg ww, respectively. The selected zinc TRV of 27 mg/kg ww, which was calculated as the fifth percentile of LOAELs based on fish data, is within the concentration range of the NOAEL and LOAEL reported for crayfish.

3.3.2.6 Tributyltin

Five studies with marine invertebrates reported tissue concentrations of TBT associated with adverse effects (Borgmann et al. 1996; Meador 1997; Meador and Rice 2001; Moore et al. 1991; Widdows and Page 1993) (Table 3-15). The LOAELs for effects on growth and reproduction ranged from 2,360 to 5,440 µg/kg dw (472 to 1,088 µg/kg ww). The NOAELs ranged from 970 to 3,960 µg/kg dw (194 to 792 µg/kg ww). Because the majority of these studies were conducted with marine species there is uncertainty with applying these values to a freshwater system. However, the

selected TBT TRV of 49.9 µg/kg ww, calculated as the product of AWQC multiplied by a BCF, is approximately four times lower than the lowest NOAEL and therefore very conservative.

3.3.2.7 PAHs

A limited number of studies were available in the literature reporting the toxicity of individual or total PAHs to benthic invertebrates (Table 3-16). Three studies with mollusks evaluated tissue concentrations of individual PAHs or a mixture of PAHs associated with adverse effects (Borchert et al. 1997; Eertman et al. 1995; Roper et al. 1997). No effects on survival were observed in clams with a tissue concentration of 1,250 µg/kg ww benzo(a)pyrene (Borchert et al. 1997). Similarly, no effects on survival were observed in zebra mussels with tissue PAH concentrations ranging from 5,600 to 10,400 µg/kg ww (Roper et al. 1997). However, it should be noted that the zebra mussel is a highly tolerant test species. Adverse effect on reproduction (abnormal gametogenesis) was reported in blue mussels exposed to aqueous fluoranthene and benzo(a)pyrene for 4 weeks (Eertman et al. 1995). The reported whole-body tissue-residue LOAELs for fluoranthene and benzo(a)pyrene were 222 and 302 µg/kg ww. The selected TRVs of 1,000 µg/kg ww for six individual PAHs, which were calculated as the product of AWQC multiplied with a BCF, were within the concentration range of LOAELs and NOAELs observed for invertebrate species. Similarly, the selected TRV of 1,000 µg/kg ww for total PAHs, which was based on benzo(a)pyrene as surrogate, was also within the concentration range of LOAELs and NOAELs observed for invertebrate species.

3.3.2.8 Phthalates

A limited number of studies were available in the literature reporting the toxicity of phthalates to benthic invertebrates (Table 3-16). Hobson et al (1984) reported no mortality of panaeid shrimps at a concentration of 18,300 µg/kg ww after exposure to di(2-ethylhexyl) phthalate through diet. Other studies, summarized in Brown and Thompson (1982) as cited in Dillon (1984), reported no effects on reproduction in crustaceans after exposure to phthalates in water at tissue concentrations ranging from 320 to 26,800 µg/kg ww. The selected TRV for dibutyl phthalate (270 µg/kg ww) calculated as the product of AWQC multiplied with a BCF was just below the range of NOAELs observed in crustaceans. The selected TRV for BEHP (390 µg/kg ww), calculated as the fifth percentile of all aquatic species LOAEL data in the Ecological PRE (Windward 2005), was in the lower range of the observed NOAELs for crustaceans.

3.3.2.9 PCBs

Four studies reported decapod tissue concentrations associated with effects after exposure to PCB Aroclor mixtures (Duke et al. 1970; Hansen et al. 1974; Nimmo et al. 1974; Sanders and Chandler 1972) (Table 3-17). These studies reported reduced survival in shrimp, crab, and crayfish species. LOAELs ranged from 1,100 µg/kg ww for mortality of grass shrimp (Hansen et al. 1974) to 42,000 µg/kg ww for mortality of brown shrimp (Hansen et al. 1974). At the lowest LOAEL, Hansen et al. (1974)

reported 33% mortality in grass shrimp following 96 hours of exposure to Aroclor 1016, compared to 8% mortality in the control group. NOAELs ranged from 1,220 µg/kg ww[20] for mortality of crayfish (Sanders and Chandler 1972) to 23,000 µg/kg ww for mortality of blue crab (Duke et al. 1970). No NOAEL was reported below the lowest LOAEL of 1,100 µg/kg ww. Five studies reported tissue concentrations associated with the toxicity of PCB Aroclors to mollusks (Boese et al. 1995; Duke et al. 1970; Hansen et al. 1974; Lowe et al. 1972; Peterson et al. 1994). Observed adverse effects included reduced shell and body growth in marine oysters. LOAELs ranged from 4,000 µg/kg ww for growth of American oysters (Hansen et al. 1974) to 119,000 µg/kg ww for growth of Eastern oysters (Lowe et al. 1972). At the lowest LOAEL, Hansen et al. (1974) reported 10% reduction in shell growth (compared with 0% in the control group) in American oysters following 96 hours of aqueous exposure to 0.6 µg/L Aroclor 1016. NOAELs ranged from 900 µg/kg ww for mortality of Asiatic clams (Peterson et al. 1994) to 101,000 µg/kg ww for growth of Eastern oysters (Lowe et al. 1972). The two NOAELs (900 and 1,750 µg/kg ww[21]) reported below the lowest LOAEL were based on accumulation studies that were not designed to measure toxicity. Based on these nine studies with benthic invertebrates the selected TRV for total PCBs of 720 µg/kg ww is a conservative value, lower than all NOAELs reported for benthic invertebrates with a factor ranging from 1.3 to 32.

3.3.2.10 DDTs

One study in the literature evaluated adverse effects to benthic invertebrates associated with tissue concentrations of 4,4′-DDD (Lotufo et al. 2000) (Table 3-18). Lethal doses that cause the death of 50% of an exposed population (LC50s) were calculated for two amphipods, Hyalella azteca and Diporeia spp. after exposed to 4,4′-DDD for 10 or 28 days. The LC50s ranged from 15,000 to 116,000 µg/kg ww. The selected TRV for 4,4′-DDD of 54 µg/kg ww, which was calculated as the product of AWQC and a BCF, is lower than concentrations associated with adverse effects to amphipods with a factor ranging from 278 to 2,148 (two orders of magnitude).

No studies in the literature evaluated adverse effects to benthic invertebrates associated with tissue concentrations of total DDTs. Three studies with decapods reported tissue concentrations of DDT associated with adverse effects (Johnson et al. 1971; Leffler 1975; Nimmo et al. 1970) (Table 3-18). These studies reported the effects of DDT on survival of decapods. LOAELs ranged from 60 µg/kg ww for mortality of pink shrimp (Nimmo et al. 1970) to 200 µg/kg ww for mortality of blue crab (Leffler 1975). At the lowest LOAEL of 60 µg/kg ww, 30% mortality, compared to 17% in the control group. was reported in pink shrimp following a 56-day exposure to a DDT mixture (Nimmo et al. 1970). There is some uncertainty associated with the concentrations reported in Nimmo et al. (1970) because shrimp were field-collected, and background concentrations in the tissues prior to DDT exposure were not reported. In addition, high mortality (17%) was reported in the control group.

[20] Concentration was converted to wet weight by assuming 74% moisture in crayfish.
[21] Concentrations were converted to wet weight assuming 80% moisture in the organism.

NOAELs ranged from 26 µg/kg mg/kg ww for mortality of blue crab (Leffler 1975) to 46 µg/kg ww[22] for mortality of crayfish (Johnson et al. 1971).

One study assessed adverse effects to amphipods from exposure to 4,4′-DDT (Lotufo et al. 2000). LC50s were calculated for *Hyalella azteca* and *Diporeia* spp. after they were exposed to 4,4′-DDT for 10 or 28 days. The LC50s ranged from 2,100 to 30,000 µg/kg ww. The study also found that 4,4′-DDT is the most toxic isomer to amphipods indicating that a direct comparison of adverse effects between total DDTs and 4,4′-DDT may not be appropriate. The relative proportion among tissue concentrations in *Hyalella azteca* causing 50 % mortality after 10 days exposure was 1:8:63 for 4,4′-DDT, 4,4′-DDD, and 4,4′-DDE. Similarly, the relative proportion among tissue concentrations in *Diporeia* spp. causing 50 % mortality after 28 days exposure was 1:6 for 4,4′-DDT and 4,4′-DDD. Low mortality was seen in exposure to 4,4′-DDE and a LC50 could not be calculated. Only one study reported the toxicity of DDT to bivalves (Binelli et al. 2001). A significant reduction in oocyte development was observed in field-collected zebra mussels with average tissue concentration of 27 µg/kg ww 4,4′-DDT. Spawning behavior changes were also observed at this LOAEL. Oocyte development was not significantly reduced in field-collected mussels with tissue concentrations ranging from 16 to 21 µg/kg ww 4,4′-DDT. The selected TRV of 290 µg/kg ww for total DDTs was calculated as the fifth percentile of all aquatic species LOAEL data in the Ecological PRE (Windward 2005). The reported LOAELs and NOAELs for the isomer 4,4′-DDT and DDTs span a large range of concentrations (approximately two orders of magnitude). The selected TRV for total DDTs falls within this range of concentrations.

3.3.2.11 beta-HCH

A literature search found no toxicological studies with mollusks that reported tissue concentrations of beta-HCH associated with adverse effects (Windward 2005).

3.3.2.12 Endrin

Only two studies with invertebrates reported tissue concentrations of endrin associated with adverse effects (Table 3-18) (Tyler-Schroeder 1979; Mason and Rowe 1976). Tyler-Schroeder measured effects on growth, mortality, and reproduction in grass shrimp following aqueous exposure to endrin. LOAELs ranged from 48 µg/kg ww for reproduction of grass shrimp to 176 µg/kg ww for mortality of grass shrimp. NOAELs ranged from 48 µg/kg ww for growth of grass shrimp to 80 µg/kg ww for mortality of grass shrimp. No NOAEL that was lower than the lowest LOAEL of 48 µg/kg ww was reported. Mason and Rowe reported mortality in eastern oysters following 1 week of aqueous exposure to endrin. The reported LOAEL and NOAEL were 16,400 and 26 µg/kg ww, respectively, when 90 and 0% mortality were observed, respectively. The selected TRV for endrin of 25 µg/kg ww is a conservation value, lower than the reported NOAELs for shrimp and oysters, with a factor ranging from 1.9 to 10.

[22] Converted to dw from ww assuming 74% moisture in crayfish.

3.3.3 Summary

Tissue exposure point concentrations (EPCs) presented in Section 3.3.1 were compared to aquatic TRVs presented in Section 3.3.2 to derive tissue hazard quotients (HQs) for all of the benthic invertebrate tissue Round 2 COPCs: antimony, arsenic, cadmium, copper, zinc, TBT, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, pyrene, total PAHs, BEHP, dibutyl phthalate, total PCBs, 4,4′-DDD, total DDTs, beta-HCH, and endrin. The HQ results for the benthic invertebrate tissue-residue assessment are presented in Section 3.6.1.2.

3.4 NEAR-BOTTOM SURFACE WATER ASSESSMENT

The exposure of and effects on the benthic community related to surface water were assessed by comparing near-bottom water chemistry concentrations to AWQC or TRVs available in the literature. This LOE was also used in the risk assessment for shellfish and crayfish.

The initial screening process for the near-bottom surface water assessment is presented in Attachment G2. The screening process included selection of COIs and identification of Round 2 COPCs by comparing the maximum concentrations measured in near-bottom surface water samples to Eco SLs. Ten Round 2 COPCs were identified based on near-bottom surface water samples collected with either XAD or peristaltic pump (Table 3-19). Figure 3-3 presents all of the near bottom surface water sampling locations.

3.4.1 Round 2 COPC Exposure Assessment

This section presents the exposure assessment of 10 Round 2 COPCs identified in near-bottom surface water data for benthic invertebrates. This assessment was conducted to provide a general understanding of the overall risk estimates to benthic invertebrates from near-bottom surface water across the Study Area and across seasonal changes.

Round 2 sampling activities at the 17 near-bottom sampling locations involved three surface water sampling events (i.e., November 2004, March 2005, and July 2005) to capture the seasonality of the water flow for the LWR (Integral 2004). Surface water sampling methods are described in Section 2.1.3.

The Round 2 COPCs and their concentrations are summarized in Table 3-20. Tables 3-21 and 3-22 presents the chemical concentrations for the 10 identified Round 2 COPCs at the 17 near-bottom sampling locations. Total PCB concentrations are represented as the sum of PCB congeners for XAD samples and as the sum of PCB Aroclors for peristaltic samples. The potential exposure of benthic invertebrates to the 10 Round 2 COPCs are summarized below.

Zinc –Dissolved zinc was detected in all sampling locations during at least one sampling event.

Benzo(a)anthracene – Benzo(a)anthracene was detected during at least one sampling event at all sampling locations, except the following: downstream of Oregon Steel Mills (W001), adjacent to Sauvie Island (W002), at the mouth of Saltzman Creek (W017), adjacent to Gunderson (W019), at Swan Island Lagoon (W021), and at Balch Creek Cove (W022).

Benzo(a)pyrene – Benzo(a)pyrene was detected during at least one sampling event at the following eight near-bottom sampling locations: at the mouth of Multnomah Channel (W003), at the head of Slips 1 and 3 in Terminal 4 (W007 and W008), adjacent to Gasco (W012), at Willamette Cove (W013), downstream of and adjacent to Arkema (W015 and W016), and at Swan Island Lagoon (W018).

4-Chloro-3-methylphenol – 4-Chloro-3-methylphenol was not detected in 43 of 55 (89%) surface water samples collected using the peristaltic method. Chloro-3-methylphenol was detected during the November 2004 or March 2005 sampling events at the following five near-bottom sampling locations: at the mouth of Multnomah Channel (W003), near the west bank at RM 4 (W006), at the head of Slip 1 in Terminal 4 (W007), at Willamette Cove (W013), and at Swan Island Lagoon (W018).

Total PCBs –Total PCBs were not detected in 34 of 53 (64%) surface water samples that were collected using the peristaltic method. Total PCBs were detected during at least one sampling event at the following six near-bottom sampling locations: downstream of Oregon Steel Mills (W001), at the head of the International Slip (W004), at Willamette Cove (W013), downstream of and adjacent to Arkema (W015 and W016), and at Swan Island Lagoon (W018).

2,4′-DDD – 2,4′-DDD was not detected in the peristaltic samples but was detected in all XAD samples collected at the following four near-bottom surface water locations: at Willamette Cove (W013), downstream of and adjacent to Arkema (W015 and W016), and at Swan Island Lagoon (W018).

2,4′-DDT – 2,4′-DDT was not detected in the peristaltic samples except one NJ-qualified concentration downstream of Oregon Steel Mills (W001). 2,4′-DDT was detected in all but one of the XAD samples, which were collected at following four near-bottom surface water locations: at Willamette Cove (W013), downstream of and adjacent to Arkema (W015 and W016), and at Swan Island Lagoon (W018).

4,4′-DDD – 4,4′-DDD was not detected in the peristaltic samples except one sample collected in July at location W012. 4,4′-DDD was detected in all XAD samples collected at the following four near-bottom surface water locations: at Willamette Cove (W013), downstream of and adjacent to Arkema (W015 and W016), and at Swan Island Lagoon (W018).

4,4′-DDT – 4,4′-DDT was not detected in the peristaltic samples except two samples collected in July (W017) and March (W019). 4,4′-DDT was detected in all XAD samples collected at the following four near-bottom surface water locations: at Willamette Cove (W013), downstream of and adjacent to Arkema (W015 and W016), and at Swan Island Lagoon (W018).

Total DDTs – Total DDTs were not detected in the peristaltic samples except four samples collected at the following near-bottom locations: downstream of Oregon Steel Mills (W001 [NJ-qualified data]), adjacent to Gasco (W012), at the mouth of Saltzman Creek (W017), and adjacent to Gunderson (W019). Total DDTs were detected in all XAD samples collected at the following four near-bottom surface water locations: at Willamette Cove (W013), downstream of and adjacent to Arkema (W015 and W016), and at Swan Island Lagoon (W018).

Surface water exposure concentrations for benthic invertebrates were also represented by the UCL of the mean concentration over all near-bottom surface water samples collected from within the Study Area, including both peristaltic pump and XAD samples. The UCL is a conservative estimate of the average concentration of Round 2 COPCs in surface water and assumed to be representative of the exposure of benthic invertebrates to chemicals in surface water. UCLs were derived using ProUCL software. Prior to running data through ProUCL to determine UCLs, non-detected data were adjusted to one-half the detection limit to represent the concentration value. Attachment G1 presents the summary statistics (i.e., minimum, maximum, and mean Round 2 COPC concentrations), distribution types, ProUCL-recommended UCLs, and selected surface water exposure concentrations for Round 2 COPCs.

3.4.2 Round 2 COPC Effects Assessment

This section identifies the studies or sources from which the Eco SLs used to identify benthic invertebrate Round 2 COPCs were derived. Eco SLs were selected based on the proposed hierarchy detailed in Attachment G3. Table 3-23 presents the chronic and acute Eco SLs for the 10 benthic invertebrate Round 2 COPCs. Brief discussions of the basis for the selection of each Round 2 COPC Eco SL follow.

Zinc – The Eco SLs for zinc, which were based on AQWC, are hardness-dependent and were adjusted from the AWQC values using EPA-provided equations (EPA 2006f) and a hardness of 25 mg/L calcium carbonate (as estimated for LWR). Criteria are for the dissolved fraction of zinc, which is the bioavailable fraction.

Benzo(a)anthracene and benzo(a)pyrene – The Eco SLs for these two PAHs were based on Tier II values (Suter and Tsao 1996) derived from 96-hour LC50 daphnid (*Daphnia magna*) studies. No toxicity data were available specifically for benthic invertebrates and the Eco SLs may therefore over or underestimate risks.

4-Chloro-3-methylphenol – The acute Eco SLs for 4-chloro-3-methylphenol was obtained from ODEQ's Water Quality Guidance Values Summary (Table 33), which

can be used in applying Oregon's Narrative Toxics Criteria (Oregon Administrative Rules [OAR] 340-041-0033(1)) to waters of the state in order to protect aquatic life (ODEQ 2001). The source for this acute criterion is uncertain. In accordance with EPA's comments (EPA 2006d), an uncertainty factor (UF) of 50 was used to extrapolate a chronic screening value from the acute Eco SL because no chronic data were available; however, there is high uncertainty in the use of a UF of 50.

Total PCBs – The acute Eco SL for total PCBs was based on the value (or benchmark) developed by ODEQ and is the same as the Tier II value presented in Suter and Tsao (1996). The acute Eco SL was calculated for AWQC, although it has never been adopted. The chronic Eco SL was based on ODEQ based on partial life-cycle tests conducted with three invertebrate and two fish species. Suter and Tsao cited the same chronic value for total PCBs and noted that it was based on fish residual values.

DDTs – The Eco SLs for 2,4′-DDD, 4,4′-DDD, 2,4′-DDT, 4,4′-DDT, and total DDTs are based on the acute and chronic AWQC for 4,4′-DDT.

3.5 TRANSITION ZONE WATER ASSESSMENT

Risks to the benthic community from TZW were assessed by comparing TZW chemistry concentrations to AWQC or TRVs available in the literature. This LOE was also used in the risk assessment for shellfish and crayfish. The initial screening process for the TZW assessment is presented in Attachment G2. The screening process included selection of COIs and identification of Round 2 COPCs by comparing the maximum concentrations measured in TZW samples to Eco SLs. Fifty-three Round 2 COPCs were identified. This included 8 metals, 2 herbicides (dalapon and Silvex™), 16 PAHs, 6 pesticides (2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDE, 4,4′-DDT, and total DDTs), 3 SVOCs (1,2-dichlorobenzene, 1,4-dichlorobenzene, and dibenzofuran), and 16 VOCs, perchlorate, and cyanide (Table 3-24).

Round 2 COPCs were evaluated using the TZW framework to identify the list of potential iCOCs in TZW for benthic invertebrates (Section 3.5.1). The exposure and effects assessments of the potential TZW iCOCs are presented in Sections 3.5.2 and 3.5.3, respectively.

3.5.1 Round 2 COPC Evaluation

Round 2 COPCs in TZW were further evaluated using the TZW framework to narrow the list of COPCs and identify chemicals that may potentially pose risks to benthic invertebrates via TZW. The chemicals that were retained following further evaluation through the TZW framework were identified as potential iCOCs for TZW. A summary of the TZW framework is presented in Figure 3-4. Potential iCOCs in TZW were integrated across other LOEs to identify iCOCs for benthic invertebrates in the risk conclusions (Section 3.8).

3.5.1.1 Transition Zone Water Evaluation Framework

A TZW framework was developed to identify potential iCOCs in TZW for ecological receptors (Figure 3-4). All data collected in the biologically active zone (≤ 38 cm) were included in assessment of TZW, including unfiltered and filtered Trident, peepers, and unfiltered Geoprobe® data. In addition, offshore groundwater sampling was performed by Siltronic in May and June of 2005 using the Geoprobe® sampling method, and those data were evaluated in the TZW framework.

3.5.1.1.1 Development of Ecological Screening Levels

Eco SLs were used both in the screening assessment (Attachment G3) and in the TZW assessment for benthic invertebrates to evaluate potential risks. Eco SLs were developed through a review of water quality benchmarks and literature-based thresholds (Attachment G3). Eco SLs were developed for all TZW COIs except individual dioxins and furans, because no data were available for the individual dioxin and furan COIs in TZW.

For metals, Eco SLs were hardness adjusted, when appropriate. The toxicity of some metals depends upon the hardness of the water. Filtered TZW samples had measured average hardness equivalent to 478 mg/L $CaCO_3$, a median of 238 mg/L $CaCO_3$, and a maximum of 3,357 mg/L $CaCO_3$ based on 93 samples. The Eco SLs for hardness-dependent metals were modified using the following EPA National Recommended Water Quality Criteria (EPA 2006f) formula for hardness:

$$\text{CCC (dissolved)} = \exp\{m_c[\ln(\text{hardness})] + b_c\} \qquad \textbf{Equation 3-1}$$

Where:

CCC = criterion continuous concentration (an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed indefinitely without resulting in an unacceptable effect)

m_c = constant that varies by metal

b_c = constant that varies by metal

Eco SLs for metals were also adjusted, if the criteria were developed using dissolved concentrations, and then compared to dissolved concentrations, when appropriate.

3.5.1.1.2 Determination of Groundwater Sources

In the TZW framework, groundwater sources of Round 2 COPCs were determined by assessment of potential groundwater pathways from upland groundwater plumes to the transition zone within the Study Area. Comparisons between chemicals produced and chemicals found in TZW samples were used to determine if complete pathways for groundwater sources were evident (Integral 2006). Integrated analyses of discharge and sampling results, seepage meter results, Trident probe temperature measurements, sediment texture, stratigraphic information, and TZW chemistry in conjunction with available site data were also used to determine if a complete groundwater pathway was apparent (Integral 2006).

Nine properties were evaluated for evidence of complete pathways for transport of upland groundwater contaminants to the transition zone (Figure 3-5). Two distinct sections were analyzed at the Arkema site because the property contained an acid plant and a chlorate plant in separate locations with potentially different groundwater characteristics (Integral 2006). The nine properties are identified as:

- Arco Terminal 22T
- Arkema (acid plant and chlorate plant)
- ExxonMobil Oil terminal
- Gasco
- Gunderson
- Kinder Morgan Linnton Terminal
- Rhône-Poulenc
- Siltronic
- Willbridge bulk fuels terminals

If a complete groundwater pathway was verified in the Portland Harbor RI/FS, the Round 2 COPCs associated with the source were carried forward for in the TZW framework. Round 2 COPCs with an incomplete groundwater pathway were not retained for further analysis.

3.5.1.1.3 Evaluation of Metals

The metals identified as Round 2 COPCs were evaluated in the TZW framework by comparing upstream sediment concentrations (for those Round 2 COPCs where upstream data were available) to Study Area sediment concentrations. If the maximum metal concentration did not the upstream sediment concentration, the metal Round 2 COPC was not retained for further analysis.

3.5.1.1.4 Additional Round 2 COPC Evaluation

Round 2 COPCs were also evaluated in the TZW framework, based on additional information, including: 1) a spatial evaluation of TZW concentration exceedances of Eco SLs and an evaluation of metal Round 2 COPC sediment concentration trends across the Study Area (by river mile), 2) an evaluation of the magnitude of TZW exceedances of Eco SLs, and 3) an evaluation of the how the dilution factor between TZW and near-bottom surface water data and porewater ventilation would effect the benthic invertebrate exposure concentration of Round 2 COPCs.

3.5.1.2 Results of the TZW Framework Evaluation

Round 2 COPCs were further evaluated using the TZW framework to identify potential iCOCs in TZW (Table 3-25). The results of this evaluation are presented in the following subsections.

3.5.1.2.1 Round 2 COPC Groundwater Source Evaluation

Based on the groundwater source evaluation step in the TZW pathway, one Round 2 COPC (i.e., chloroethane) was eliminated from further analysis. Complete groundwater pathways were identified at the Arkema and Siltronic properties. A potential groundwater source was identified at the Rhône-Poulenc property, and an indeterminate pathway was identified at the Gasco property. No definitive groundwater pathways were established at ARCO, ExxonMobil Oil, Gunderson, Kinder Morgan Linnton, or Willbridge (Integral 2006). Non-metal analytes at the sites with incomplete groundwater pathways were removed from further consideration as TZW potential iCOCs. The only non-metal analyte excluded from further consideration based on an incomplete groundwater pathway was chloroethane.

3.5.1.2.2 Round 2 COPC Metals Comparison to Upstream Data

Based on the comparison of metals concentration to upstream sediment, no Round 2 COPCs in TZW were eliminated from further analysis. When data were available, Round 2 metal COPCs (i.e., cadmium, lead, copper, and zinc) were compared to upstream background concentrations in sediment, to evaluate metals Round 2 COPCs in TZW with a potential sediment source. The maximum concentration of each of these metal Round 2 COPCs exceeded upstream concentrations in sediment. The percent exceedance of TZW concentrations of the upstream sediment UCL on 90th percentile threshold level ranged from 36% (cadmium) to 62% (lead) (Table 3-26).

3.5.1.2.3 Additional Round 2 COPC Evaluation

All remaining Round 2 COPCs were further evaluated in the TZW framework by analyzing site-wide trends in TZW and exceedances of Eco SLs across the Study Area. Hazard quotients (HQs) were derived and evaluated to determine the spatial extent of TZW concentrations exceeding Eco SLs. HQs were calculated by dividing the maximum TZW concentration for Round 2 COPCs in each TZW sample by the chronic Eco SL. Based on the analysis of spatial extent of TZW concentrations, additional Round 2 COPCs in TZW were eliminated from further analysis.

Metals

Based on the analysis of Study Area-wide trends of TZW concentrations, three metal Round 2 COPCs in TZW (i.e., barium, sodium, and vanadium) were eliminated from further analysis. Sodium and vanadium have elevated sediment concentrations in limited areas of the Study Area. Sodium concentrations in sediment spike between RM 7.05 and RM 7.56. Vanadium concentrations in TZW exceeded Eco SLs at the Siltronic site, and concentrations in sediment samples exhibited a corresponding increase in the vicinity of the same site. The limited spatial extent of elevated sodium and vanadium concentrations reduces their relevance as population-level stressors. Barium concentrations in sediment were consistent across river miles. Because of the ubiquitous presence of this metal in sediment, barium concentrations in TZW are considered representative of the anthropogenic background level in an urban area. Therefore, barium, sodium, and vanadium were not carried forward as potential iCOCs for TZW.

In addition, based on the analysis of spatial extent and magnitude of TZW exceedances, five metal Round 2 COPCs in TZW (i.e., cadmium, copper, lead, nickel, and zinc) were eliminated from further analysis. Cadmium, copper, lead, and nickel all have mean HQs less than 1.0. Copper exceeds the chronic Eco SL at one site, while these four metals have low exceedance frequencies. The zinc TZW concentration exceedance of the chronic Eco SL was based on one detected sample. The limited spatial extent and magnitude of the TZW concentration exceedances of Eco SLs of these five metal Round 2 COPCs reduces their relevance as population level stressors.

Herbicides

Based on the analysis of spatial extent of TZW concentrations, the two herbicide Round 2 COPCs in TZW (i.e., dalapon and Silvex™) were eliminated from further analysis. TZW concentrations of dalapon and Silvex™ both have a low magnitude of exceedance (i.e., all maximum HQs < 10); maximum chronic HQs for dalapon and Silvex™ are of 1.2 and 4.4, respectively at one site (Rhône-Poulenc). In addition, Silvex™ has a low detection frequency; Silvex™ concentrations were detected in 1 of 15 samples.

SVOCs

Based on the analysis of spatial extent of TZW concentrations, all three SVOC Round 2 COPCs in TZW (i.e., dibenzofuran, 1,2-dichlorobenzene, and 1,4-dichlorobenzene) were eliminated from further analysis. Dibenzofuran was excluded from the list of potential iCOC for TZW because of a low magnitude of exceedance(i.e., all maximum HQs < 10) and low detection frequencies. Dibenzofuran TZW concentrations exceeded the chronic and acute Eco SLs at two sites, Siltronic and Gasco, with chronic maximum HQs of 2.0 and 2.2, respectively. 1,2-Dichlorobenzene and 1,4-dichlorobenzene had limited distribution of elevated TZW concentrations; these two SVOCs exceeded chronic and acute Eco SLs at one site (Rhône-Poulenc). 1,2-dichlorobenzene and 1,4-dichlorobenzene were not identified as a Round 2 COPC in other LOE. Thus, these three SVOCs were eliminated as potential iCOCs for TZW.

VOCs

None of the 16 VOC Round 2 COPCs were identified as potential iCOCs for TZW due to: 1) the low magnitude of exceedances, 2) the limited spatial extent of exceedances, and, 3) because of the effect of the dilution factor between TZW and near-bottom surface water data and porewater ventilation on TZW exposure concentrations. There were low magnitude of exceedances (i.e., maximum HQs < 10) of Eco SLs for the eight following VOCs: 1,1-dichloroethene, 1,2,4-trimethylbenzene, 1,3,5-trimethylbenzene, chloroethane, isopropylbenzene, m,p-xylene, trichloroethene, and vinyl chloride. There was also a limited spatial extent of elevated Round 2 COPC concentrations (i.e., the chemical exceeded the chronic and acute Eco SLs in a limited number of samples or sites) for the following

eight VOCs: benzene, carbon disulfide, chlorobenzene, cis-1,2-dichloroethene, ethylbenzene, o-xylene, toluene, and total xylenes.

In addition, proximal near-bottom water concentrations of all VOCs identified as Round 2 COPCs were up to five orders of magnitude lower than concentrations in TZW, indicating dilution between TZW and surface water. Therefore, a 10% porewater ventilation assumption of TZW by benthic organisms, (also used in the food web model FWM; see Appendix E), was applied to determine risk to benthic organisms from VOCs. The porewater ventilation fraction assumption of 1 to 10% was confirmed for all taxa observed in the benthic community survey based on a literature search. These two pieces of information were used to determine that TZW exposure concentrations of VOC Round 2 COPCs would be diluted enough result in HQs below 1.0 and therefore, are not expected to result in risks to benthic invertebrates. No VOC Round 2 COPCs were identified as potential iCOCs in TZW.

3.5.1.2.4 Identification of Potential iCOCs in TZW

The remaining chemicals not excluded as part of the TZW framework evaluation (i.e., PAHs [2-methylnaphthalene, acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenzo(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3-cd) pyrene, naphthalene, phenanthrene, and pyrene], DDTs [2,4′-DDD, 4,4′-DDD, 4,4′-DDE, 4,4′-DDT, and total DDTs], cyanide, and perchlorate) were retained as potential TZW iCOCs (Table 3-27). The exposure and effects assessment of these 24 potential TZW iCOCs are presented in the following subsections.

3.5.2 Potential TZW iCOC Exposure Assessment

This section presents the assessment of benthic invertebrate exposure to the 23 identified potential iCOCs for TZW. This assessment was conducted to provide a general understanding of the overall risk estimates to the benthic community across the Study Area.

All TZW samples, including unfiltered Trident, filtered Trident, peeper, and unfiltered Geoprobe® samples, collected in the biologically active zone (≤ 38cm) were included in the assessment of benthic invertebrates. LWG TZW samples were collected using an unfiltered Trident push probe, a filtered Trident push probe, a Geoprobe®, and small-volume peepers between October 3 and December 2, 2005. Additional offshore groundwater samples were collected by Siltronic in May and June 2005 using the Geoprobe® sampling method. Sampling locations were selected at each of the nine properties based on the results of the groundwater discharge mapping field effort (Integral 2006). TZW sampling methods are described in Section 2.1.4.

The detection frequency of 25 potential iCOCs in TZW and a general comparison of TZW concentrations to Eco SLs (described in Section 3.5.3) follow.

2-Methylnaphthalene – 2-Methylenaphthalene was detected in 27 of 93 (29%) TZW samples. Of the 27 detected samples, 11 exceeded the chronic Eco SL, and 5 exceeded the acute Eco SL. Exceedances occurred equally in all three sampling methods: Trident filtered (3), Trident and/or Geoprobe® unfiltered (4), peeper (4). No peeper samples exceeded the acute Eco SL.

Acenaphthene – Acenaphthene was detected in 100 of 106 (94%) TZW samples and exceeded the chronic Eco SL in 24 samples. The acute Eco SL was exceeded in 10 of the 24 samples. Of the 24 chronic exceedances, 15 occurred in the unfiltered Trident and/or Geoprobe®, while the remaining exceedances were divided between the filtered Trident (5) and the unfiltered peeper (4). Of the 10 acute exceedances, 9 occurred in the Trident and/or Geoprobe®, and 1 occurred in an unfiltered peeper sample.

Anthracene – Anthracene was detected in 76 of 106 (72%) TZW samples. Of these 76 detected samples, 28 exceeded the chronic Eco SL, with 18 of the exceedances in unfiltered Trident and/or Geoprobe® samples and the remaining exceedances divided equally between the filtered Trident (5) and unfiltered peeper (5) samples. Five samples exceeded the acute Eco SL in unfiltered Trident and/or Geoprobe® samples.

Benzo(a)anthracene – Benzo(a)anthracene was detected in 44 of 106 (42%) TZW samples. Of these 44 samples, 31 (70% of detected) exceeded the chronic Eco SL, and 14 exceeded the acute Eco SL. The majority of the chronic Eco SL exceedances occurred in the unfiltered Trident and/or Geoprobe® samples (25), with the remaining chronic exceedances divided between the filtered Trident (2) and peeper (4) samples. All 14 acute Eco SL exceedances occurred in the unfiltered Trident and/or Geoprobe® samples.

Benzo(a)pyrene – Benzo(a)pyrene was detected in 38 of 106 (36%) TZW samples. Of these 38 detected samples, 34 (89.5%) exceeded the chronic Eco SL, while 15 exceeded the acute Eco SL. The majority of the chronic Eco SL exceedances occurred in samples collected with the unfiltered Trident and/or Geoprobe® (30), with the remaining chronic Eco SL exceedances divided equally between the filtered Trident (2) and the unfiltered peeper (2) samples. All 15 of the acute Eco SL exceedances occurred in samples collected with the unfiltered Trident and/or Geoprobe®.

Benzo(b)fluoranthene – Benzo(b)fluoranthene was detected in 31 of 106 (29%) TZW samples. Of these 31 detected samples, 13 exceeded the chronic Eco SL, and all 13 were collected with an unfiltered Trident and/or Geoprobe®. No acute Eco SL was available for this chemical.

Benzo(g,h,i)perylene – Benzo(g,h,i)perylene was detected in 40 of 106 (38%) TZW samples. Of these 40 detected samples, 13 exceeded the chronic Eco SL, and all 13 were collected with an unfiltered Trident and/or Geoprobe®. No acute Eco SL was available for this chemical.

Benzo(k)fluoranthene – Benzo(k)fluoranthene was detected in 27 of 106 (25%) TZW samples. Of these 27 detected samples, 10 exceeded the chronic Eco SL, and all 10 were collected with an unfiltered Trident and/or Geoprobe®. No acute Eco SL was available for this chemical.

Chrysene – Chrysene was detected in 46 of 106 (43%) TZW samples. Of these 46 detected samples, 10 exceeded the chronic Eco SL in unfiltered Trident and/or Geoprobe® samples. No acute Eco SL was available for this chemical.

Dibenzo(a,h)anthracene – Dibenzo(a,h)anthracene was detected in 30 of 106 (28%) TZW samples. Of these 30 detected samples, 8 samples collected with the unfiltered Trident and/or Geoprobe® exceeded the chronic Eco SL. No acute Eco SL was available for this chemical.

Fluoranthene – Fluoranthene was detected in 67 of 106 (63%) TZW samples. Of these 67 detected samples, 11 exceeded the chronic Eco SL, and 4 exceeded the acute Eco SL. All exceedances of both the acute and chronic Eco SLs were in unfiltered Trident and/or Geoprobe® samples.

Fluorene – Fluorene was detected in 85 of 106 (80%) of TZW samples. Of the 85 detected samples, 36 exceeded the chronic Eco SL and 2 exceeded the acute Eco SL. Most of the chronic Eco SL exceedances were in unfiltered Trident and/or Geoprobe® samples (22), while the other exceedances were evenly divided between the filtered Trident (7) and peeper (7) samples. The two acute Eco SL exceedances were in unfiltered Trident and/or Geoprobe® samples.

Indeno(1,2,3-cd)pyrene – Indeno(1,2,3,-cd)pyrene was detected in 39 of 106 (37%) of TZW samples. Of these 39 detected samples, 13 samples collected with and unfiltered Trident and/or Geoprobe® exceeded the chronic Eco SL. No acute Eco SL was available for this chemical.

Naphthalene – Naphthalene was detected in 75 of 173 (43%) TZW samples. Of these 75 detected samples, 31 exceeded the chronic Eco SL, and 18 exceeded the acute Eco SL. The majority of chronic (28) and acute (15) exceedances were in unfiltered Trident and/or Geoprobe® samples.

Phenanthrene – Phenanthrene was detected in 71 of 106 (67%) TZW samples. Of these 71 detected samples, 36 exceeded the chronic Eco SL, and 13 exceeded the acute Eco SL. Most of the chronic (23) and acute (12) exceedances were in the unfiltered Trident and/or Geoprobe® samples.

Pyrene – Pyrene was detected in 72 of 106 (68%) TZW samples. Of these 72 detected samples, 11 samples collected with an unfiltered Trident and/or Geoprobe® exceeded the chronic Eco SL. No acute Eco SL was available for this chemical.

DDTs – 2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDE, 4,4′-DDT, and total DDTs were all detected in TZW samples. All DDTs exceeded the chronic Eco SL, and 4,4′-DDD, 4,4′-DDT, and total DDTs exceeded the acute Eco SL. The majority of chronic exceedances occurred in unfiltered Trident and/or Geoprobe® samples. Similarly, 4,4′-DDD, 4,4′-DDT, and total DDT acute Eco SL exceedances occurred in the unfiltered Trident and/or Geoprobe® samples only.

Cyanide – Cyanide was detected in 32 of 34 (94%) TZW samples at two sites where it was analyzed (Gasco and Siltronic). All 32 detections exceeded the chronic Eco SL, and 25 exceeded the acute Eco SL. The majority of exceedances (23) occurred in unfiltered Trident and/or Geoprobe® samples; the remainder (9) occurred in unfiltered peeper samples.

Perchlorate – Perchlorate was detected in 11 of 21 (52%) TZW samples collected at two of the three sites where it was analyzed (Arkema acid plant and Arkema chlorate plant). Perchlorate was not detected in TZW samples collected from the Gunderson site. All 11 detected concentrations exceeded the chronic Eco SL, and none exceeded the acute Eco SL. The majority of exceedances (i.e., in 9 samples) occurred in the unfiltered Trident samples; the remainder (i.e., in 2 samples) occurred in unfiltered peeper samples.

3.5.3 Potential TZW iCOC Effects Assessment

This section identifies the studies or sources from which the Eco SLs used to evaluate potential TZW iCOCs were derived. Eco SLs were selected based on the proposed hierarchy detailed in Attachment G3. Table 3-28 presents the chronic and acute Eco SLs for potential TZW iCOCs. Brief discussions of the basis for the selection of each Eco SL are presented in the subsections that follow.

2-Methlynaphthalene – The acute and chronic Eco SLs of 37 and 2.1 µg/L, respectively, are based on the Tier II values for 1-methlynaphthalene(Suter and Tsao 1996). 1-Methlynaphthalene was used as a surrogate compound and was assumed to be protective for assessing risk to 2-methlynaphthalene. These criteria are a secondary acute value (SAV) and secondary chronic value (SCV) calculated from an EC50 fathead minnow fish study.

Acenaphthene – The acute and chronic Eco SLs of 80 and 23 µg/L, respectively, are derived from an FAV and an FCV EPA calculation used for sediment quality guideline development (Suter and Tsao 1996).

Anthracene – The acute and chronic Eco SLs of 13 and 0.73 µg/L, respectively, are based on the Tier II (SAV and SCV) values calculated from an EC50 daphnid study (Suter and Tsao 1996).

Benzo(a)anthracene, and benzo(a)pyrene – The Eco SLs for these two PAHs were based on Tier II values (Suter and Tsao 1996) derived from 96-hour LC50 daphnid studies.

Benzo(b)flouranthene, benzo(g,h,i)perylene, benzo(k)flouranthene, chrysene, dibenzo(a,h)anthracene, indeno (1,2,3-cd)pyrene, and pyrene – The chronic Eco SLs for these PAHs were based on PAH-specific FCVs (EPA 2003c). The FCVs were derived from national water quality criteria (WQC) guidelines (Stephan et al. 1985). Acute Eco SLs for these PAHs were not included because the acute screening value was derived using a method than that selected for the chronic Eco SL, no acute value was available, and/or the acute screening level was lower than the chronic Eco SL.

Fluoranthene – The acute and chronic Eco SLs of 33.6 and 6.16 µg/L, respectively, are based on the Tier II values (SAV and SCV)used for sediment quality guideline development (Suter and Tsao 1996).

Fluorene – The acute and chronic Eco SLs of 70 and 3.9 µg/L, respectively, are based on the Tier II values (SAV and SCV) (Suter and Tsao 1996). These values are derived from the Office of Solid Waste and Emergency Response calculations for ecotoxicity thresholds.

Naphthalene – The acute and chronic Eco SLs of 33.6 and 6.16 µg/L, respectively, are based on the Tier II values derived from an LC50 rainbow trout study (Suter and Tsao 1996).

Phenathrene – The acute and chronic Eco SLs of 190 and 12 µg/L, respectively, are based on the Tier II values (SAV and SCV) used for sediment quality guideline development (Suter and Tsao 1996).

DDTs – The Eco SLs for 2,4′-DDD, 4,4′-DDD, 2,4′-DDT, 4,4′-DDT, and total DDTs are based on the acute and chronic AWQC for 4,4′-DDT.

Cyanide – The Eco SLs for cyanide were based on the acute and chronic AWQC.

Perchlorate – The chronic Eco SL for perchlorate of 18 µg/L is a LOAEL based on a 70-day exposure of amphibians (Goleman et al. 2002). No acute Eco SL was developed because none was found in the sources reviewed.

3.5.4 Equilibrium Partitioning Assessment

EqP calculations were performed to estimate TZW concentrations for chemicals with a potential sediment source. Two Round 2 COPCs (acenaphthene and acetone) were identified in the COI screen (Attachment G2). The maximum predicted TZW concentrations for these two Round 2 COPCs exceeded the chronic Eco SL. These Round 2 COPCs were further evaluated by comparing predicted TZW concentrations from the 95th percentile sediment concentration to Eco SLs (Table 3-29). Predicted

TZW concentrations were less than the Eco SLs for both Round 2 COPCs evaluated in the EqP evaluation.

3.6 ROUND 2 RISK RESULTS

This section presents the risk results for each LOE evaluated for benthic invertebrates. Risk results combine the exposure and effects estimates for each LOE. Results of the benthic community risk evaluation are presented in Section 3.6.1, and results of the shellfish and crayfish risk evaluation are presented in Sections 3.6.2 and 3.6.3, respectively.

3.6.1 Benthic Community

Risks to the benthic community are assessed by three measures of exposure and effect as presented in Section 3.1.4. The different LOEs associated with each measure are presented in the following sections: sediment toxicity testing and modeling (Section 3.6.1.1), tissue-residue data compared to literature-based TRVs (Section 3.6.1.2), and near-bottom surface water and TZW exposure data compared to AWQC or literature effects data (Sections 3.6.1.3 and 3.6.1.4).

3.6.1.1 Sediment Toxicity Testing and Modeling

This section presents the risk assessment to the benthic community based on sediment toxicity testing and predicted effects based on two models. Adverse effects to the benthic community were assessed at 227 locations within the Study Area and six upstream locations by conducting sediment toxicity tests with the midge, *Chironomus tentans*, and the amphipod, *Hyalella azteca*. Two endpoints, mortality and growth, were evaluated, and based on the biological effects definitions presented in Section 3.2.1 the sediments were categorized into four effects levels: no effect, not reportable, minor effects (Level 2), and moderate effects (Level 3). Table 3-30 presents the effect designation of the 233 sediment samples based on the four toxicity test endpoints, and Table 3-31 presents the number of locations and the overall effects levels.[23] Figures 3-6 and 3-7 present the results of the two toxicity tests for the 227 sediment samples in the Study Area. Individual points exceeding either Level 2 or Level 3 are found throughout the Study Area.

The second LOE, based on predicted toxicity to the benthic community, was evaluated in areas where toxicity tests were not performed. Toxicity was predicted by both models based on model-specific thresholds. For the FPM, 19 chemicals and 3 conventional parameters were identified as being associated with the observed toxicity (Table 3-32). A sample with chemical concentrations that exceeded any single Level 3 threshold was classified as predicted "toxic"; a sample with chemical concentrations all below Level 2 thresholds was classified as predicted "non-toxic"; and a sample with chemical concentrations that exceeded Level 2 but were below

[23] The proportions of hits shown in these tables do not necessarily correspond to the proportion of the area of the LWR that is affected because samples were not uniformly distributed.

Level 3 thresholds was classified as "indeterminate." Some chemicals, such as ammonia, mercury, and residual-range hydrocarbons, have different SQVs at Level 2 and Level 3. Other chemicals, such as arsenic, diesel-range hydrocarbons, and total PCBs, have the same SQV at both levels. Although at first this may appear unusual, it reflects the fact that the concentration-toxicity curve for these chemicals is apparently steep in Portland Harbor. At the level at which the effects associated with these chemicals can be reliably seen, the effect is clear enough that it exceeds both Level 2 and Level 3.

A review of the toxicity test results indicates that many of the same locations exceed both Level 2 and Level 3, which results in the pattern of site-specific SQVs observed by the FPM. From a practical standpoint, this creates a relatively clear distinction between areas that are not likely to experience effects and areas in which the benthic community may be at greater risk, without a large "grey zone" in between.

For the LRM, 18 chemicals and 2 conventional parameters were identified as associated with the observed toxicity (Table 3-32). A sample with a PrMax value > 0.61 was classified as predicted toxic; a sample with a PrMax value < 0.40 was classified as predicted non-toxic; and a sample with a PrMax value in between was classified as indeterminate. The LRM threshold for non-toxic samples was chosen as the PrMax value with 20% false negatives, a level chosen as an upper bound for prediction errors of non-toxic samples. Similarly, the LRM threshold for toxic samples was chosen as the PrMax value with approximately 20% false positives (actual error rate was 19%). The PrMax thresholds of 0.40 and 0.61 correspond to probabilities of toxicity of 0.35 and 0.60, respectively, through inversion of the logistic calibration equation for the multi-chemical model.[24] The chemicals listed in Table 3-32 are Round 2 COPCs.

EPA has requested that iAOPCs be identified as part of the Comprehensive 2 Report. In addition, EPA requested in their comments to the Benthic Interpretive Report (Windward et al. 2006) that both models, the FPM and LRM, be used to identify iAOPCs based on risks to the benthic community. LWG agreed to comply with their request for this report and developed a decision matrix (Table 3-33) that was used to identify iAOPCs based on both models (LWG 2006b). However, because of the several uncertainties associated with the LRM and the combination of the two models, iAOPCs were also identified based on the FPM alone.

The uncertainties associated with the LRM and combining the two models include:

- Irreproducibility of the results of the LRM
- Failure of the *Hyalella* growth endpoint to predict measured benthic toxicity

[24] PrMax = $1/(1+\exp(-(-1.6172 + 3.4292\ p_{max})))$, where pmax = probability of toxicity from the individual chemical models; see Step 9 of Section 5.3.1 of the Benthic Interpretive Report (Windward et al. 2006) for more information.

- Reduction in predictive accuracy

LWG has not been able to reproduce the results of the LRM as provided by NOAA because the model includes a national database that currently is unavailable (Field 2006). The inclusion of the national database affected the calculation of sums (i.e., PAHs); therefore, LWG could not derive the same values as those used in the LRM. This issue may be resolved if the database used in the model becomes available during the preparation of the BERA.

The *Hyalella* growth endpoint had substantially lower reliability as a predictor of measured benthic toxicity than did the other three endpoints (i.e., *Chironomus* mortality and growth and *Hyalella* mortality) in both the FPM and LRM (Windward et al. 2006). This was the only endpoint that was not capable of reliably predicting toxicity in Portland Harbor sediments at Levels 2 and 3. The *Hyalella* growth endpoint was correlated most strongly with percent fines and ammonia and had only weak correlations with a few metals. Pooling this endpoint with *Hyalella* mortality, which was otherwise quite reliable, also reduced the reliability of the pooled endpoint below acceptable levels. For these reasons, this endpoint, as well as the pooled *Hyalella* endpoint, should not be used in predicting iAOPCs in Portland Harbor.

The predictions of the two models as presented in this section are being combined for policy rather than technical reasons, and as a result, the combined predictive accuracy is reduced compared to either model individually. The reason for the reduced accuracy is that each model is fully optimized using its own mathematical procedures; and by combining them, the predictive accuracy of both models is degraded.

For this report, the two models were combined and the decision matrix presented in Table 3-33 was used to identify iAOPCs. For areas where no sediment toxicity testing was performed, the areas were identified as not toxic or toxic where the two models agree, or where one model produced a "conclusive" prediction (i.e., not toxic or toxic) and the other model produced an "inconclusive" prediction. In areas classified as "indeterminate," other LOEs and spatial considerations (i.e., neighboring benthic toxicity data and predictions) were used to assess whether the potential risk to the benthic community was sufficient to conclude that an area should be included in an iAOPC and whether it indicated a potential Round 3B data need. Table 3-34 presents the classification of the 1,047 surface sediment samples with chemistry data only in the ERA dataset. Of these samples, 704 were classified as not toxic, 163 as indeterminate, and 160 as toxic. Because the two models predict toxicity based on different chemicals, not all sediment samples in the database were assessed by both models. Sixteen LWG core sediment samples were not assessed by either model because the samples were analyzed primarily for conventionals. The FPM did not evaluate 39 sediment samples because these samples were analyzed for PAHs only. Uncertainties associated with the chemistry data are discussed in the uncertainty assessment (Section 3.7). The sampling locations and hit classification of the

1,031 sediment samples for each model are presented in Figures 3-8 and 3-9, and the overall hit classification based on the decision matrix is presented in Figure 3-10.

Further assessment of risks to the benthic community presented in this report is based on the FPM because of the uncertainties associated with the LRM. The chemicals associated with toxicity based on the FPM were identified as potential iCOCs (Table 3-35). The evaluation of the toxicity tests and the classification of sediment samples with chemistry data only is presented in Figure 3-8.

3.6.1.2 Tissue-Residue Assessment

This section presents the preliminary risk characterization by combining the exposure estimates (Section 3.3.1) and effects estimates (Section 3.3.2) and calculating HQs. HQs were calculated for each Round 2 COPC using the following equation:

$$HQ = \frac{EPC}{TRV}$$ **Equation 3-2**

Where:

HQ	=	hazard quotient (unitless)
EPC	=	exposure point concentration in tissue (µg/kg ww or mg/kg ww; represented by a UCL concentration)
TRV	=	aquatic toxicity reference value in tissue residue (µg/kg ww or mg/kg ww)

HQs were calculated for each individual sample and for the site-wide UCL. If the site-wide UCL was > 1.0, the COPC was identified as a potential iCOC. The potential iCOCs were evaluated further (Section 3.6.1.2.3) to identify Round 2 iCOCs for benthic invertebrates (Section 3.8.2). As stated in Section 3.3, all tissue data were used in the risk assessment of the benthic community. Field-collected and laboratory-exposed clam tissue samples were used in the risk assessment for clams and field-collected crayfish tissue samples were used in the risk assessment for crayfish.

Seventeen Round 2 COPCs were identified based on both field-collected and laboratory-exposed tissue samples. The Round 2 COPCs included four metals (arsenic, cadmium, copper, and zinc), TBT, six individual PAHs (benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, and pyrene), total PAHs, BEHP, dibutyl phthalate, total PCBs, total DDTs, and 4,4′-DDD. Ten Round 2 COPCs, antimony, arsenic, cadmium, zinc TBT, benzo(a)pyrene, pyrene, total PAHs, beta-HCH, and endrin, were identified based on the predicted tissue approach.

The field-collected tissue assessment is presented in Section 3.6.1.2.1, the laboratory tissue assessment is presented in Section 3.6.1.2.2, and potential iCOCs identified based on empirical data are presented in Section 3.6.1.2.3. The predicted tissue assessment, including potential iCOCs, is presented in Section 3.6.1.2.4.

3.6.1.2.1 Field-Collected Tissue

Three types of invertebrate tissue (clam, crayfish, and invertebrates[25] collected with multiplate samplers) were collected in the field. Nine Round 2 COPCs were identified based on field-collected clams. The range of HQs for individual samples and the UCL for all samples across the site for these Round 2 COPCs are presented in Table 3-36. The locations of field-collected clam tissue samples with HQs > 1.0 and the Round 2 COPCs are presented in Figure 3-11.

The concentrations of cadmium in clams collected at nine Round 2 locations between RM 2 and RM 4.5 and at four locations between RM 8.2 and RM 8.8 exceeded the aquatic TRV of 0.09 mg/kg ww with HQs ranging from 1.03 to 2.4. In addition, the UCL exceeded the TRV with an HQ of 1.2. All concentrations of copper measured in field-collected clam samples (Round 1 and 2) and the UCL exceeded the aquatic TRV of 3.1 mg/kg ww with HQs ranging from 2.2 to 4.4. The concentration of zinc measured in field-collected clams exceeded the aquatic TRV of 27 mg/kg ww in all Round 2 tissue samples except in samples collected at Terminal 4 upstream of Slip 3 (BT010), in Willamette Cove (BT016), and adjacent to Arkema (BT018). The HQ for the tissue samples exceeding the TRV ranged from 1.1 to 2.0, and the site-wide UCL exceeded the TRV with an HQ of 1.4. The concentrations of TBT exceeded the aquatic TRV of 49.9 µg/kg ww in two tissue samples collected in the navigation channel at the mouth of the International Slip (BT005) and adjacent to Portland Shipyard (BT023) with HQs of 1.3 and 11. In addition, the site-wide UCL exceeded the TRV with an HQ of 4.4.

The concentrations of total PAHs measured in field-collected clams exceeded the aquatic TRV of 1,000 µg/kg ww at four locations: downstream of Arco (BT012), US Moorings embayment (BT014), adjacent to Gasco (BT015), and downstream of Arkema (BT017) with HQs ranging from 1.0 to 5.0. The UCL for total PAHs did not exceed the TRV. One detected concentration of dibutyl phthalate measured in clams collected in Swan Island Lagoon adjacent to the Coast Guard (BT022) exceeded the aquatic TRV of 270 µg/kg ww with an HQ of 4.8. In addition, the UCL exceeded the TRV with an HQ of 1.9. The concentration of total PCBs measured in field-collected clams exceeded the aquatic TRV of 720 µg/kg ww in one tissue sample collected in Willamette Cove (BT016) with an HQ of 3.7. The UCL for total PCBs did not exceed the TRV.

The tissue concentrations of 4,4′-DDD measured in field-collected clams exceeded the aquatic TRV of 54 µg/kg ww at two Round 2 locations, downstream of or adjacent to Arkema (BT017 and BT018), and at one Round 1 location also near Arkema (07R006). The HQ for the three tissue samples exceeding the TRV ranged from 1.6 to 3.0. In addition, the UCL exceeded the TRV with an HQ of 1.5. The concentrations of total DDTs measured in field-collected clams exceeded the aquatic TRV of 290 µg/kg ww at one Round 2 location, downstream of Arkema (BT017),

[25] Epibenthic invertebrates and zooplankton.

and at one Round 1 location also near Arkema (07R006). The HQ for the two tissue samples exceeding the TRV was 1.6. The UCL for total DDTs did not exceed the TRV.

Copper was the only Round 2 COPC identified based on crayfish tissue samples collected in Round 1. All tissue concentrations of copper exceeded the aquatic TRV of 3.1 mg/kg ww with HQs ranging from 3.4 to 5.7. In addition, the UCL exceeded the TRV with an HQ of 4.8. Similarly, copper was the only Round 2 COPC identified based on invertebrates collected by multiplate samplers. Two tissue samples were analyzed for copper and one of these two samples collected near Oregon Steel Mills (MIT001) exceeded the TRV with an HQ of 1.9 (Table 3-37). The locations of field-collected crayfish and invertebrate tissue samples with HQs > 1.0 and the Round 2 COPC are presented in Figure 3-11.

3.6.1.2.2 Laboratory-Exposed Tissue

Two types of invertebrate tissue (clam and worms) were exposed in the laboratory to sediments collected in the Study Area. Six Round 2 COPCs including copper, TBT, total PAHs, BEHP, total DDTs and 4,4′-DDD were identified based on laboratory-exposed clams. The range of HQs for individual samples and the site-wide HQs based on the UCL for these Round 2 COPCs are presented in Table 3-38. The locations of sediment samples used in the laboratory exposure of clams with HQs > 1.0 and the COPCs are presented in Figure 3-12.

The concentrations of copper in all but one laboratory-exposed clam tissue sample exceeded the TRV of 3.1 mg/kg ww with HQs ranging between 1.0 and 1.9. The tissue samples not exceeding the TRV were composed of clams exposed to sediment collected at the mouth of Saltzman Creek (BT020). In addition, the UCL exceeded the TRV with an HQ of 1.3. The concentration of TBT in one laboratory clam tissue sample exposed to sediment collected at Portland Shipyard (BT023) exceeded the aquatic TRV of 49.9 µg/kg ww with an HQ of 14. The site-wide UCL exceeded the TRV with an HQ of 4.2. The total PAHs concentration in one laboratory clam tissue sample exposed to sediment collected downstream of Arco (BT012) exceeded the aquatic TRV with an HQ of 1.3. The site-wide UCL for total PAHs did not exceed the TRV. The concentration of BEHP in one laboratory clam tissue sample exposed to sediment collected adjacent to Gunderson (BT028) exceeded the aquatic TRV with an HQ of 22. The UCL exceeded the TRV with an HQ of 3.6. All other concentrations of BEHP in laboratory clam tissue samples were qualified as either undetected (U) or estimated (J). Concentration of 4,4′-DDD measured in two tissue samples exposed to sediment collected downstream of or adjacent to Arkema (BT017 and BT018) exceeded the aquatic TRV of 54 µg/kg ww with HQs of 1.6 and 13. In addition, the UCL exceeded the TRV with an HQ of 4.1. The concentration of total DDTs measured in one tissue sample exposed to sediment collected adjacent to Arkema (BT018) exceeded the aquatic TRV of 290 µg/kg ww with an HQ of 3.6. The UCL exceed the TRV with an HQ of 1.1.

Sixteen Round 2 COPCs including four metals (arsenic, cadmium, copper, and zinc), TBT, six individual PAHs (benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, and pyrene), total PAHs, dibutyl phthalate, total PCBs, total DDTs and 4,4′-DDD were identified based on laboratory-exposed worms. The range of HQs for individual samples and the site-wide HQs based on the UCL for these Round 2 COPCs are presented in Table 3-39. The locations of sediment samples used in the laboratory exposure of worms with HQs > 1.0 and the Round 2 COPCs are presented in Figure 3-12.

The concentration of arsenic in laboratory worms exposed to sediments collected at seven locations in the Study Area exceeded the TRV of 1.7 mg/kg ww with HQs ranging from 1.0 to 1.8. These locations were distributed throughout the Study Area and included the navigation channel at the mouth of the International Slip (BT005), in Terminal 4 Slip 1 (BT008), adjacent to Linnton Plywood (BT011), adjacent to Gasco (BT015), in Reidell Cove (BT019), near McCall upstream of Willbridge docks (BT021), and adjacent to Goldendale Aluminum (BT033). The UCL for arsenic did not exceed the TRV. The concentration of cadmium in laboratory worms exposed to sediments collected at four locations in the Study Area exceeded the TRV of 0.09 mg/kg ww with HQs ranging from 1.0 to 2.8. These four sampling stations were located between RM 2.3 and RM 4.2 and included upstream of Oregon Steel Mill (BT002), near Time Oil (BT004), head of the International Slip (BT006), and near the property line between the International Slip and Terminal 4 (BT007). The UCL for cadmium did not exceed the TRV.

The concentrations of copper in laboratory worms exposed to sediments collected at four locations in the Study Area exceeded the TRV of 3.1 mg/kg ww with HQs ranging from 1.1 to 6.5. These locations were included the head of the International Slip (BT006), near Cathedral Park (BT013), adjacent to Portland Shipyard (BT023), and in Swan Island Lagoon (BT026). The UCL exceeded the TRV with an HQ of 1.2. The concentration of zinc in laboratory worms exposed to sediments collected at 12 locations in the Study Area exceeded the TRV of 27 mg/kg ww with HQs ranging from 1.0 to 1.2. These locations were distributed throughout the Study Area from RM 2 to RM 10. The UCL for zinc did not exceed the TRV. The concentration of TBT in one tissue sample of worms exposed to sediment collected at Portland Shipyard (BT023) exceeded the aquatic TRV of 49.9 µg/kg ww with an HQ of 34. In addition, the UCL exceeded the TRV with an HQ of 11.

Individual PAH TRVs of 1,000 µg/kg ww were exceeded by six individual PAHs (benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, and pyrene) measured in laboratory worms exposed to sediments collected at one or all of the following locations: downstream of Arco (BT012), in the US Moorings embayment (BT014), adjacent to Gasco (BT015), and downstream and adjacent to Arkema (BT017 and BT018). The UCLs for three of these PAHs (benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene) did not exceed the TRV. The UCL for benzo(a)anthracene barely exceeded the TRV with an HQ of 1.0,

and the UCLs for chrysene and pyrene exceeded the TRV with HQs of 1.7 and 4.6, respectively. The concentrations of total PAHs measured in laboratory-exposed worms exposed to sediments collected at 13 locations in the Study Area exceeded the TRV of 1,000 µg/kg ww with HQs ranging from 1.0 to 37. These locations were distributed throughout the Study Area from RM 4.4 to RM 9.5. In addition, the UCL exceeded the TRV with an HQ of 16.

The concentration of dibutyl phthalate in one laboratory worm tissue sample exposed to sediment collected at the mouth of Saltzman Creek (BT020) exceeded the aquatic TRV of 270 µg/kg ww with an HQ of 1.7. This concentration of dibutyl phthalate was J-qualified. The UCL for dibutyl phthalate did not exceed the TRV.

The concentrations of total PCBs measured in laboratory worms exposed to sediments collected at six locations in the Study Area exceeded the aquatic TRV of 720 µg/kg ww with HQs ranging from 1.0 to 6.0. These locations were distributed throughout the Study Area from RM 2.3 to RM 9.7 and include near Oregon Steel Mills (BT002), in Willamette Cove (BT016), downstream of Arkema (BT017), in Swan Island Lagoon (BT026), adjacent to Gunderson (BT028), and in Balch Creek Cove (BT032). The UCL for total PCBs exceeded the TRV with an HQ of 3.3.

The concentrations of 4,4′-DDD measured in laboratory worms exposed to sediments collected at five locations in the Study Area exceeded the aquatic TRV of 54 µg/kg ww with HQs ranging from 1.1 to 20. These locations were downstream of Arco (BT012), in the US Moorings embayment (BT014), downstream and adjacent to Arkema (BT017 and BT018), and adjacent to Gunderson (BT028). In addition, concentrations of total DDTs measured in laboratory worms exposed to sediments collected downstream and adjacent to Arkema (BT017 and BT018) exceeded the aquatic TRV of 290 µg/kg ww with HQs of 3.5 and 5.1, respectively. Both UCLs exceeded the TRVs with HQs of 7.8 and 2.1, respectively.

3.6.1.2.3 Identification of Potential iCOCs

Round 2 COPCs were identified as potential iCOCs if the site-wide UCL HQ > 1.0 in any of the five tissue types (clams, crayfish, or multiplate invertebrates collected in the field and laboratory-exposed clams and worms). Only one chemical, copper, was identified as a Round 2 COPC in invertebrate tissue samples collected with multiplate samplers based on maximum concentration exceeding the aquatic TRV. Because only two tissue samplers were analyzed for copper, a UCL was not calculated. The UCLs were used because they were more conservative than the 80th percentile (i.e., HQs based on UCLs were greater than HQs based on 80th percentiles). The 80th percentile concentration is generally recommended to evaluate a population-level endpoint when exposure varies across individuals in the population. The following 12 chemicals were identified as potential iCOCs for the benthic community.

Cadmium – A site-wide UCL HQ of 1.2 for cadmium was calculated based on field-collected clam tissue samples. The selected aquatic TRV of 0.09 mg/kg ww used in the initial screening process is a conservative value calculated as the fifth

percentile of all aquatic species LOAEL data in the Ecological PRE (Windward 2005). This value is lower than all NOAELs reported for benthic invertebrates with a factor ranging from 5 to 5,900 (approximately 6 orders of magnitude) (Section 3.3.2). A comparison of cadmium concentrations in the Study Areas to upstream concentrations was performed to evaluate if the concentrations in the Study Area was elevated. The evaluation found that 36% of the cadmium concentrations in the Study Area exceeded the UCL on the 90th percentile of upstream cadmium concentrations. Urban runoff has been found to be an important source of cadmium in urban receiving waters (Walker et al. 1999).

Copper – The site-wide UCL HQs for both field-collected tissue and laboratory-exposed tissue samples ranged from 1.2 to 4.8. The selected aquatic TRV of 3.1 mg/kg ww used in the initial screening process is a conservative value calculated as the fifth percentile of LOAELs based on fish data. This value is lower than all NOAELs reported for benthic invertebrates with a factor ranging from 2 to 16 (Section 3.3.2). However, it should be noted that all invertebrate studies evaluated adverse effects on survival. A comparison of copper concentrations in the Study Areas to upstream concentrations was performed to evaluate if the concentrations in the Study Area was elevated. The evaluation found that 46% of the copper concentrations in the Study Area exceeded the UCL on the 90th percentile of upstream copper concentrations. Urban runoff has been found to be an important source of copper in urban receiving waters (Walker et al. 1999).

Zinc – A site-wide UCL HQ of 1.4 for zinc was calculated based on field-collected clam tissue samples. The selected aquatic TRV of 27 mg/kg ww used in the initial screening process is calculated as the fifth percentile of LOAELs based on fish data and is within the concentration range of the NOAEL and LOAEL reported in one study with crayfish (Section 3.3.2). A comparison of zinc concentrations in the Study Areas to upstream concentrations was performed to evaluate if the concentrations in the Study Area was elevated. The evaluation found that 39% of the zinc concentrations in the Study Area exceeded the UCL on the 90th percentile of upstream zinc concentrations. Urban runoff has been found to be an important source of zinc in urban receiving waters (Walker et al. 1999).

TBT – The site-wide UCL HQs for field-collected clam tissue samples and laboratory-exposed clam and worm tissue samples ranged from 4.2 to 11. The selected aquatic TRV of 49.9 µg/kg ww used in the initial screening process is a conservative value calculated as the product of AWQC multiplied with a BCF. This value is approximately 4 times lower than the lowest NOAEL reported for marine invertebrates. A less conservative TRV of 3 mg/kg dw (600 µg /kg ww assuming 80% moisture) has been used at other sediment clean up sites in the Pacific Northwest (Meador et al. 2002). If this TRV is used in the HQ calculations, the UCL HQ for field-collected clam tissue, laboratory clam tissue, and laboratory worm tissue sample would range between 0.35 and 0.90. Because the UCL HQs < 1.0 for all invertebrate tissue types based on this TRV, TBT will not be carried forward as an iCOC.

Chrysene – A site-wide UCL HQ of 1.7 for chrysene was calculated based on laboratory worm tissue samples. The UCL HQ > 1.0 was driven by three tissue samples exposed to sediments collected downstream of Arco, at US Moorings, and adjacent to Gasco. The selected aquatic TRV of 1,000 µg/kg ww used in the initial screening process is calculated as the product of AWQC multiplied with a BCF. A limited number of studies were available in the literature reporting the toxicity of individual or mixture of PAHs to benthic invertebrates. The selected TRV for chrysene is within the concentration range of LOAELs and NOAELs observed for individual and mixture of PAHs and invertebrate species.

Pyrene – A site-wide UCL HQ of 4.6 for pyrene was calculated based on laboratory worm tissue samples. The UCL HQ > 1.0 was driven by five tissue samples exposed to sediments collected downstream of Arco, at US Moorings, adjacent to Gasco, and downstream and adjacent to Arkema. The selected aquatic TRV of 1,000 µg/kg ww used in the initial screening process is calculated as the product of AWQC multiplied with a BCF. A limited number of studies were available in the literature reporting the toxicity of individual or mixture of PAHs to benthic invertebrates. The selected TRV for pyrene is within the concentration range of LOAELs and NOAELs observed for individual and mixture of PAHs and invertebrate species.

Total PAHs – A site-wide UCL HQ of 16 for total PAHs was calculated based on laboratory worm tissue samples. The UCL HQ > 1.0 was driven by 13 tissue samples exposed to sediments collected throughout the Study Area. The selected aquatic TRV of 1,000 µg/kg ww used in the initial screening process was based on using benzo(a)pyrene as surrogate. A limited number of studies were available in the literature reporting the toxicity of individual or mixture of PAHs to benthic invertebrates. The selected TRV for total PAHs is within the concentration range of LOAELs and NOAELs observed for individual and mixture of PAHs and invertebrate species.

BEHP – A site-wide UCL HQ of 3.6 for BEHP was calculated based on laboratory clam tissue samples. The UCL HQ > 1.0 was driven by one tissue sample (8,600 µg/kg) exposed to sediments collected adjacent to Gunderson. The detected concentrations of BEHP in the other laboratory-exposed tissue samples ranged from 53 to 160 µg/kg (mean detected concentration including the high concentration was 410 µg/kg). BEHP was detected in the exposure sediment at a concentration of 1,000 µg/kg dw. The concentration of BEHP in other sediment samples collected in the vicinity (within approximately 150 ft) of the laboratory exposure sediment sample was heterogeneous with concentrations ranging from 166 to 4,500 µg/kg. The selected aquatic TRV of 390 µg/kg ww used in the initial screening process is calculated as the fifth percentile of all aquatic species LOAEL data in the Ecological PRE (Windward 2005). A limited number of studies were available in the literature reporting the toxicity of BEHP and other phthalates to crustaceans. The selected TRV for BEHP is in the lower range of the observed NOAELs for crustaceans. Because the

high tissue concentration was an outliner indicating a highly limited spatial exposure BEHP will not be carried forward as an iCOC.

Dibutyl phthalate – A site-wide UCL HQ of 1.9 for dibutyl phthalate was calculated based on field-collected clam tissue samples. The UCL HQ > 1.0 was driven by one tissue sample exposed to sediments collected in Swan Island Lagoon adjacent to the Coast Guard. Dibutyl phthalate was not detected in the other field-collected tissue samples. In addition, dibutyl phthalate was not detected in the co-located sediment sample (4.65 µg/kg dw). The concentration of dibutyl phthalate in other sediment samples collected in the vicinity (within approximately 150 ft) of the co-located sediment sample had concentrations ranging from 5 to 90 µg/kg dw. The selected aquatic TRV of 270 µg/kg ww used in the initial screening process is calculated as the product of AWQC multiplied with a BCF. A limited number of studies were available in the literature reporting the toxicity of BEHP and other phthalates to crustaceans. The selected TRV for dibutyl phthalate is just below range of the observed NOAELs for crustaceans. Because the high tissue concentration was an outliner indicating a highly limited spatial exposure dibutyl phthalate will not be carried forward as an iCOC.

Total PCBs – A site-wide UCL HQ of 3.3 for total PCBs was calculated based on laboratory worm tissue samples. The UCL HQ > 1.0 was driven by six tissue samples exposed to sediments collected adjacent to Oregon Steel Mills, in Willamette Cove, downstream of Arkema, in Swan Island Lagoon, adjacent to Gunderson, and in Balch Creek Cove. The selected aquatic TRV of 720 µg/kg ww used in the initial screening process is a conservative value calculated as the fifth percentile of all aquatic species LOAEL data in the Ecological PRE (Windward 2005). This value is lower than all NOAELs reported for benthic invertebrates with a factor ranging from 1.3 to 32.

4,4′-DDD – The site-wide UCL HQs for field-collected clam tissue samples and laboratory-exposed clam and worm tissue samples ranged from 1.5 to 7.8. The UCL HQ > 1.0 for field-collected clams was driven by three tissue samples collected adjacent to or downstream of Arkema (including one Round 1 sample). The UCL HQ > 1.0 for laboratory-exposed clams was driven by two tissue samples exposed to sediments collected adjacent to or downstream of Arkema. The UCL HQ > 1.0 for laboratory-exposed worms was driven by five tissue samples exposed to sediments collected downstream of Arco, adjacent to US Moorings, downstream or adjacent to Arkema, and adjacent to Gunderson. The selected aquatic TRV for 4,4′-DDD of 54 µg/kg ww is a conservative value calculated as the product of AWQC and a BCF. Only one study was available in the literature reporting the toxicity of 4,4′-DDD to amphipods. The selected TRV for 4,4′-DDD is lower than concentrations associated with adverse effects to amphipods with a factor ranging from 278 to 2,148 (two orders of magnitude).

Total DDTs – Site-wide UCL HQs of 1.1 and 1.2 were calculated based on laboratory-exposed worm and clam tissue samples. The UCL HQ > 1.0 for

laboratory-exposed clams was driven by one tissue sample exposed to sediments collected adjacent to Arkema. The UCL HQ > 1.0 for laboratory-exposed worms was driven by two tissue samples exposed to sediments collected adjacent to or downstream of Arkema. The selected aquatic TRV for total DDTs of 290 µg/kg ww is was calculated as the fifth percentile of all aquatic species LOAEL data in the Ecological PRE (Windward 2005). No studies in the literature evaluated adverse effects to benthic invertebrates associated with tissue concentrations of total DDTs. The reported LOAELs and NOAELs for the isomer 4,4′-DDT span a large range of concentrations (approximately two order of magnitude). The selected TRV for total DDTs falls within this range of concentrations.

3.6.1.2.4 Predicted Tissue

The Round 2 COPCs identified through the predictive tissue approach were further evaluated to identify potential iCOCs. For each of the 11 COPCs the site-wide UCL sediment concentration was multiplied by the BSAF compared to the aquatic TRV. If the predicted site-wide UCL tissue concentration exceeded the TRV the COPC was identified as a potential iCOCs (Table 3-40). The UCLs were used because they were more conservative than the 80th percentile (i.e., HQs based on UCLs were greater than HQs based on 80th percentiles). The 80th percentile concentration is generally recommended to evaluate a population-level endpoint when exposure varies across individuals in the population. The following nine chemicals were identified as potential iCOCs for the benthic community.

Cadmium – A predicted site-wide UCL tissue concentration of 1.04 mg/kg ww was calculated based on field-collected clams exceeding the selected aquatic TRV of 0.09 mg/kg ww with a HQ of 12. The TRV, calculated as the fifth percentile of all aquatic species LOAEL data in the Ecological PRE (Windward 2005), is a conservative value, lower than all NOAELs reported for benthic invertebrates with a factor ranging from 6 and to 5,900 (approximately 6 orders of magnitude).

Copper – A predicted site-wide UCL tissue concentration of 3.97 mg/kg ww was calculated based on laboratory-exposed clams exceeding the selected aquatic TRV of 3.1 mg/kg ww with a HQ of 1.3. The TRV, calculated as the fifth percentile of LOAELs based on fish data, is a conservative value, lower than all NOAELs reported for benthic invertebrates by a factor ranging from 2 to 16. However, although all invertebrate studies evaluated adverse effects on survival, none included sublethal effects.

Zinc – A predicted site-wide UCL tissue concentration of 35.68 mg/kg ww was calculated based on field-collected clams exceeding the selected aquatic TRV of 27 mg/kg ww with a HQ of 1.3. The TRV, calculated as the fifth percentile of LOAELs based on fish data, is within the concentration range of the NOAEL and LOAEL reported for crayfish.

TBT – A predicted site-wide UCL tissue concentration of 2.51 mg/kg lipid, 8.8 mg/kg lipid, and 7.2 mg/kg lipid was calculated based on field-collected clam,

laboratory-exposed clam, and laboratory-exposed worm tissue samples. The selected aquatic TRV of 49.9 µg/kg ww used in the initial screening process is a conservative value calculated as the product of AWQC multiplied with a BCF. This value converts to 2.2 mg/kg lipid for field-collected clams assuming 80% moisture content and 2.28% average lipid content, to 5.0 for laboratory-exposed clams assuming 80% moisture content and 1.00% average lipid content, and to 2.1 mg/kg lipid for laboratory-exposed worms assuming 80% moisture content and 2.38% average lipid content. The calculated HQs were 1.1 for field-collected clams, 1.8 for laboratory-exposed clams, and 3.4 for laboratory-exposed worms. The selected aquatic TRV is approximately 4 times lower than the lowest NOAEL reported for marine invertebrates. A less conservative TRV of 3 mg/kg dw has been used at other sediment clean up sites in the Pacific Northwest (Meador et al. 2002). This value converted to 26.3 mg/kg lipid for field-collected clams assuming 80% moisture content and 2.28% average lipid content, to 60.0 mg/kg lipid for laboratory-exposed clams assuming 80% moisture content and 1.00% average lipid content, and to 25.2 mg/kg lipid for laboratory-exposed worms assuming 80% moisture content and 2.38% average lipid content. If this TRV is used the predicted UCL for field-collected clam and laboratory-exposed clam and worm issue concentrations would not exceed the TRV. Because the UCL HQs are < 1.0 for all invertebrate tissue types based on this TRV, TBT will not be carried forward as an iCOC.

Benzo(a)pyrene – A predicted site-wide UCL tissue concentration of 176.2 mg/kg lipid was calculated based on laboratory-exposed worms exceeding the selected aquatic TRV of 1,000 µg/kg ww (42.0 mg/kg lipid assuming 80% moisture and 2.38% average lipid content) with a HQ of 4.2. The selected TRVs, calculated as the product of AWQC multiplied with a BCF, is within the concentration range of LOAELs and NOAELs observed for PAHs and invertebrate species.

Pyrene – A predicted site-wide UCL tissue concentration of 137.6 mg/kg lipid and 158.5 mg/kg lipid was calculated based on laboratory-exposed clams and worms. The selected aquatic TRV of 1,000 µg/kg ww converted to 100 mg/kg lipid for laboratory-exposed clams assuming 80% moisture content and 1.00% average lipid content and to 42.0 mg/kg lipid for laboratory-exposed worms assuming 80% moisture content and 2.38% average lipid content. The calculated HQs were 1.4 for laboratory-exposed clams and 3.8 for laboratory-exposed worms. The selected TRVs, calculated as the product of AWQC multiplied with a BCF, is within the concentration range of LOAELs and NOAELs observed for PAHs and invertebrate species.

Total PAHs – A predicted site-wide UCL tissue concentration of 122.7 mg/kg lipid, 412.1 mg/kg lipid, and 411.4 mg/kg lipid was calculated based on field-collected clam, laboratory-exposed clam, and laboratory-exposed worm tissue samples. The selected aquatic TRV of 1,000 µg/kg ww used benzo(a)pyrene as a surrogate which was calculated as the product of AWQC multiplied with a BCF. This value converted to 43.9 mg/kg lipid for field-collected clams assuming 80% moisture content and 2.28% average lipid content, to 100 mg/kg lipid for laboratory-exposed clams

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assuming 80% moisture content and 1.00% average lipid content, and to 42.0 mg/kg lipid for laboratory-exposed worms assuming 80% moisture content and 2.38% average lipid content. The calculated HQs were 2.8 for field-collected clams, 4.1 for laboratory-exposed clams, and 9.8 for laboratory-exposed worms. The selected TRVs, which used benzo(a)pyrene as surrogate, is within the concentration range of LOAELs and NOAELs observed for PAHs and invertebrate species.

beta-HCH – A predicted site-wide UCL tissue concentration of 0.68 mg/kg lipid was calculated based on laboratory-exposed clams exceeding the selected aquatic TRV of 4.9 µg/kg ww (0.49 mg/kg lipid for laboratory-exposed clams assuming 80% moisture content and 1.00% average lipid content) with a HQ of 1.4. A literature search found no toxicological studies with mollusks that reported tissue concentrations of beta-HCH associated with adverse effects (Windward 2005). Because the HQ exceedance was driven by non-detects (i.e., one-half DLs of non-detected sediment concentrations were used to calculate the sediment UCL) and NJ-qualified data, beta-HCH was not carried forward as a iCOC.

Endrin – A predicted site-wide UCL tissue concentration of 5.59 mg/kg lipid and 1.97 mg/kg lipid was calculated based on laboratory-exposed clams and worms. The selected aquatic TRV of 25 µg/kg ww converted to 2.5 mg/kg lipid for laboratory-exposed clams assuming 80% moisture content and 1.00% average lipid content and to 1.05 mg/kg lipid for laboratory-exposed worms assuming 80% moisture content and 2.38% average lipid content. The calculated HQs were 2.3 for laboratory-exposed clams and 1.9 for laboratory-exposed worms. The TRV, calculated as the fifth percentile of LOAELs based on fish data, is a conservative value, lower than all NOAELs reported for benthic invertebrates by a factor ranging from 2 to 656. Because the HQ exceedance is driven by non-detects (i.e., one-half DLs of non-detected sediment concentrations were used to calculate the sediment UCL) and NJ-qualified data endrin was not carried forward as a iCOC.

3.6.1.3 Near-Bottom Surface Water Assessment

This section presents the risk characterization estimates by combining the exposure estimates (Section 3.4.1) and effects estimates (Section 3.4.2) and calculating HQs. As stated in Section 3.4 all near-bottom surface water data are used in the risk assessments of the benthic community, shellfish, and crayfish. HQs were derived for each near-bottom surface water Round 2 COPC using the following equation:

$$HQ = \frac{EPC}{EcoSL}$$ **Equation 3-3**

Where:

HQ	=	exceedance factor hazard quotient (unitless)
EPC	=	exposure point concentration in surface water (mg/L or µg/L)
Eco SL	=	acute or chronic ecological screening level (mg/L or µg/L)

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HQs were calculated for each individual sample and for the site-wide UCL. If the site-wide UCL was > 1.0, the COPC was identified as a potential iCOC. The potential iCOCs were further evaluated (Section 3.6.1.3.2) to identify iCOCs for benthic invertebrates (Section 3.8.2).

3.6.1.3.1 Evaluation of Round 2 COPCs

Ten Round 2 COPCs were identified in the surface water screen: zinc, four organics (benzo(a) anthracene, benzo(a)pyrene, 4-chloro-3-methylphenol, and total PCBs), and five pesticides (2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDT, and total DDTs). Table 3-41 presents the chronic and acute HQs for each of the Round 2 COPCs at the 17 near-bottom sampling locations in the three surface water sampling events. Figure 3-13 present the chronic HQs for the 10 Round 2 COPCs. Brief discussions of these 10 surface water Round 2 COPCs follow.

Zinc – The surface water zinc concentration exceeded the acute and chronic Eco SLs in only one near-bottom sample, collected in Balch Creek Cove (W022) during the fall sampling event (November 2004), with HQs of 1.2 and 1.1, respectively.[26] The remaining near-bottom surface water samples had zinc HQs < 1.0.

Benzo(a)anthracene and benzo(a)pyrene – These chemicals exceeded the chronic Eco SL in only one sample collected near Gasco (W012). The chronic Eco SL for benzo(a)anthracene was exceeded in the summer sampling event with an HQ of 4.1, and the chronic Eco SL for benzo(a)pyrene was exceeded in the fall and summer sampling events (November 2004 and July 2005) with HQs of 1.4 and 11.

4-Chloro-3-methyphenol – 4-Chloro-3-methyphenol exceeded the chronic Eco SL with an HQ of 1.1 in only one sample collected at the mouth of Multnomah Channel (W003) during the spring sampling event (March 2005).

Total PCBs – Total PCBs exceeded the chronic Eco SL with an HQ of 1.2 in only one sample collected at the head of the International Slip (W004) during the spring sampling event (March 2005).

2,4′-DDD – 2,4′-DDD exceeded the chronic Eco SL in two samples collected downstream and adjacent to Arkema (W015 and W016). At W015, the Eco SL was exceeded in the fall and spring sampling events (November 2004 and March 2005), with HQs of 2.1 and 1.1, respectively. At W016, the Eco SL was exceeded in the summer sampling event (July 2005), with an HQ of 1.4.

2,4′-DDT – 2,4′-DDT exceeded the chronic Eco SL with an HQ of 19 in only one sample collected near Oregon Steel Mills (W001) during the spring sampling event (March 2005). The exceedance was based on NJ-qualified data.

[26] For zinc, the acute Eco SL is lower than the chronic Eco SL.

4,4′-DDD – 4,4′-DDD exceeded the chronic Eco SL at two locations downstream and adjacent to Arkema (W015 and W016). At W015, the chronic Eco SL was exceeded in all three sampling events, with HQs of ranging from 2.3 to 3.1. At W016, the chronic Eco SL was exceeded during the summer sampling event (July 2005), with an HQ of 3.3.

4,4′-DDT – 4,4′-DDT exceeded the chronic Eco SL at three locations downstream and adjacent to Arkema (W015 and W016) and adjacent to Gunderson (W019). At W015 the Eco SL was exceeded in the fall and spring sampling events (November 2004 and March 2005), with HQs of 1.6 and 1.1, respectively. At W016, the Eco SL was exceeded in samples collected during the fall and summer sampling event (November 2004 and July 2005), with HQs of 1.3 and 3.9, respectively. At W019, the Eco SL was exceeded in samples collected during the spring sampling event (March 2005), with an HQ of 1.2.

Total DDTs – Total DDTs exceeded the chronic Eco SL at one or all of the sampling events at the following six locations: downstream of Oregon Steel Mills (W001), adjacent to Gasco (W012), downstream of and adjacent to Arkema (W015 and W016), at the mouth of Saltzman Creek (W017), and adjacent to Gunderson (W019). The chronic HQs ranged from 1.2 to 9.8 at all locations except downstream of Oregon Steel Mills (W001), which had an HQ of 20 based on NJ-qualified data.

3.6.1.3.2 Identification of Potential iCOCs

Round 2 COPCs were identified as potential iCOCs for the benthic community, shellfish, and crayfish if the site-wide UCL HQ >1.0 in near-bottom surface water samples. The UCLs were used because they were more conservative than the 80th percentile (i.e., HQs based on UCLs were greater than HQs based on 80th percentiles). The 80th percentile concentration is generally recommended to evaluate a population-level endpoint when exposure varies across individuals in the population. The following six chemicals were identified as potential iCOC: benzo(a)pyrene, 2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDT, and total DDTs (Table 3-42).

3.6.1.4 Transition Zone Water Assessment

This section presents the risk characterization estimates by combining the exposure estimates (Section 3.5.2) and effects estimates (Section 3.5.3) and calculating HQs. As stated in Section 3.5, the TZW data were used in the risk assessment of the benthic community and shellfish. The results of the HQ calculations for the 24 potential TZW iCOCs are presented in Table 3-42 and summarized below.

2-Methylnaphthalene – 2-Methylnaphthalene exceeded the chronic Eco SL at two sites (Gasco and Siltronic) with HQs of 40 and 17, respectively. The chronic HQs ranged from 0.0016 to 40 across all sites. 2-Methylnaphthalene was analyzed at six sites.

Acenaphthene – Chronic HQs for acenaphthene ranged from 0.00010 to 17, and concentrations in TZW exceeded the chronic Eco SL at two sites (Gasco and

Siltronic), with HQs of 5.2 and 17, respectively. Chronic HQs were less than 1.0 at the four other sites where acenaphthene was analyzed.

Anthracene – Chronic HQs for anthracene ranged from 0.0018 to 87, and concentrations in TZW exceeded the chronic Eco SL at two sites (Gasco and Siltronic) with HQs (based on the maximum concentration) of 13 and 87, respectively. Chronic HQs were less than 1.0 at the four other sites where anthracene was analyzed.

Benzo(a)anthracene – Chronic HQs for benzo(a)anthracene ranged from 0.089 to 1,200, and concentrations exceeded the chronic Eco SL at five sites (Arco, ExxonMobil Oil, Gasco, Kinder Morgan, and Siltronic). Benzo(a)anthracene did not exceed the chronic SL at the Willbridge site.

Benzo(a)pyrene – Chronic HQs for benzo(a)pyrene ranged from 0.13 to 2,700, and concentrations in TZW exceeded the chronic Eco SL at four sites (Arco, ExxonMobil Oil, Gasco, and Siltronic). At those sites, the maximum chronic HQs ranged from 15 to 2,700. Benzo(a)pyrene was analyzed at two other sites (Kinder Morgan and Willbridge), but HQs at those locations were less than 1.0.

Benzo(b)fluoranthene – Chronic HQs for benzo(b)fluoranthene ranged from 0.0032 to 49, and concentrations in TZW exceeded the chronic Eco SL at two (Gasco and Siltronic) of the six sites. At the four other sites where benzo(b)fluoranthene was analyzed, chronic HQs were less than 1.0.

Benzo(g,h,i)perylene – Chronic HQs for benzo(g,h,i)perylene ranged from 0.0093 to 66, and concentrations in TZW exceeded the chronic Eco SL at three sites (ExxonMobil Oil, Gasco, and Siltronic). At the three other sites where benzo(g,h,i)perylene was analyzed, chronic HQs were less than 1.0.

Benzo(k)fluoranthene – Chronic HQs for benzo(k)fluoranthene ranged from 0.0023 to 14, and concentrations in TZW exceeded the chronic Eco SL at two sites (Gasco and Siltronic) with HQs (based on the maximum concentrations) of 3.1 and 14, respectively. At the four other sites where benzo(k)fluoranthene was analyzed, chronic HQs were less than 1.0.

Chrysene – Chronic HQs for chrysene ranged from 0.00069 to 17, and concentrations in TZW exceeded the chronic Eco SL at two sites (Gasco and Siltronic). At the four other sites (Arco, ExxonMobil Oil, Kinder Morgan, and Willbridge) where chrysene was analyzed, HQs were less than 1.0.

Dibenzo(a,h)anthracene – Chronic HQs for dibenzo(a,h)anthracene ranged from 0.0064 to 13, and concentrations in TZW exceeded the chronic Eco SL at two sites (Gasco and Siltronic). At the four other sites (Arco, ExxonMobil Oil, Kinder Morgan, and Willbridge) where dibenzo(a,h)anthracene was analyzed, HQs were less than 1.0.

Fluoranthene – Chronic HQs for fluoranthene ranged from 0.0011 to 17, and concentrations in TZW exceeded the chronic Eco SL at two (Gasco and Siltronic) of six sites. The highest HQ (17.2) occurred at the Siltronic site. HQs were less than 1.0 at the four other sites where it was analyzed (Arco, ExxonMobil Oil, Kinder Morgan, and Willbridge).

Fluorene – Chronic HQs for fluorene ranged from 0.00079 to 28, and concentrations in TZW exceeded the chronic Eco SL at three sites (ExxonMobil Oil, Gasco, and Siltronic). At the three other sites where fluorene was analyzed (Arco, Kinder Morgan, and Willbridge), HQs were less than 1.0.

Indeno(1,2,3-cd) pyrene – Chronic HQs for indeno(1,2,3-cd) pyrene ranged from 0.0084 to 61, and concentrations in TZW exceeded the chronic Eco SL at three sites (ExxonMobil Oil, Gasco, and Siltronic). At the three other sites where indeno(1,2,3-cd) pyrene was analyzed (Arco, Kinder Morgan, and Willbridge), HQs were less than 1.0.

Naphthalene – Chronic HQs for naphthalene ranged from 0.00053 to 1,100, and concentrations in TZW exceeded the Eco chronic SL at three sites (the Arkema acid plant, Gasco, and Siltronic). At the seven other sites where naphthalene was analyzed, the HQs were less than 1.0.

Phenanthrene – Chronic HQs for phenanthrene ranged from 0.00063 to 57, and concentrations in TZW exceeded the chronic Eco SL at three sites (ExxonMobil Oil, Gasco, and Siltronic). At the three other sites where phenanthrene was analyzed (Arco, Kinder Morgan, and Willbridge), the HQs were less than 1.0.

Pyrene – Chronic HQs for pyrene ranged from 0.00098 to 15, and concentrations in TZW exceeded the chronic Eco SL at two sites (Gasco and Siltronic). At the four other sites where pyrene was analyzed (Arco, ExxonMobil Oil, Kinder Morgan, and Willbridge), the HQs were less than 1.0.

DDTs – Chronic HQs for DDTs (i.e., 2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDE, 4,4′-DDT, and total DDTs) were greater than 1.0 at two sites (the Arkema acid plant and Rhône-Poulenc). DDTs were analyzed in TZW at these two sites only. Chronic HQs for 2,4′-DDD ranged from 3.3 to 1,100; chronic HQs for 2,4′-DDT ranged from 0.89 to 150; chronic HQs for 4,4′-DDD ranged from 4.7 to 1,300, chronic HQs for 4,4′-DDE ranged from 3.9 to 930; chronic HQs for 4,4′-DDT ranged from 5.0 to 1,800; and chronic HQs for total DDTs ranged from 7.7 to 3,100,.

Cyanide – Chronic HQs for cyanide ranged from 1.2 to 4,400; the mean HQ was 280. Cyanide exceeded the chronic SL in TZW at the two sites where it was analyzed, Gasco and Siltronic. TZW sampled from Gasco had the highest concentration (23.1 mg/L).

Perchlorate – Chronic HQs for perchlorate ranged from 0.22 to 9,800; the mean HQ was 280. Perchlorate exceeded the chronic SL in TZW at the two of the three sites where it was analyzed (at the Arkema acid plant and Arkema chlordate plant). HQs were less than 1.0 in all TZW samples from Gunderson.

The two Round 2 COPCs evaluated in the EqP approach, acenaphthylene and acetone, were not carried forward as a potential TZW iCOC because the predicted TZW concentration based on 95th percentile sediment concentrations (Table 3-29) were less then the Eco SLs. Chronic HQs for acenaphthylene and acetone were 0.024 and 0.15, respectively.

3.6.2 Shellfish

Risks to shellfish were assessed based on the following LOEs: field-collected and laboratory-exposed clam tissue data compared to aquatic TRVs, predicted tissue data compared to aquatic TRVs, using the *Hyalella* and *Chirnomus* toxicity test results as surrogates for effects to bivalves, and empirical surface water and TZW data compared to AWQC or TRVs.

3.6.2.1 Tissue-Residue Assessment

The exposure of and effects on shellfish were assessed by comparing tissue-residue concentrations to TRVs available in the literature. The tissue-residue data were derived using field-collected and laboratory-exposed clams, which were presented as two LOEs in Table 3-2. Because the risk assessment for clams based on field-collected and laboratory tissue was presented as part of the risk assessment for the benthic community only a brief summary is included below. The Round 2 COPC exposure assessment is presented in Section 3.3.1, the Round 2 COPC effects assessment in Section 3.3.2, and the risk estimates in Section 3.6.1.2.

Six potential iCOCs were identified based on field-collected clams, five potential iCOCs were identified based on laboratory-exposed clams, and eight potential iCOCs were identified based on predicted tissue concentrations for field-collected and laboratory-exposed clams. The six potential iCOC for field-collected clams were cadmium, copper, zinc, TBT, dibutyl phthalate, and 4,4′-DDD. The five potential iCOCs for laboratory-exposed clams were copper, TBT, BEHP, 4,4′-DDD, and total DDTs. The eight potential iCOCs based on predicted tissue concentrations were cadmium, copper, zinc, TBT, pyrene, total PAHs, beta-HCH, and endrin. TBT will not be carried forward as an iCOC because the aquatic TBT TRV was very conservative and HQs were < 1.0 based on a TRV of 600 µg/kg ww (Meador et al. 2002). In addition, BEHP and dibutyl phthalate will not be carried forward as iCOCs because the single high tissue concentration for each chemical was an outliner, indicating a highly limited spatial exposure (see Section 3.6.1.2.3).

3.6.2.2 Sediment Toxicity Test Results

The exposure of and effects on shellfish related to chemicals in sediment were assessed using the results of the toxicity tests with *Chironomus tentans* and *Hyalella azteca* as surrogates for potential effects to shellfish.

Two endpoints, mortality and growth, were evaluated in the toxicity tests and based on the biological effects definitions presented in Section 3.2.1 the sediments were categorized into three effects levels: no effect, minor effects (Level 2), and moderate effects (Level 3). Figures 3-6 and 3-7 present the results of the two toxicity tests for the 227 sediment samples in the Study Area. Individual points exceeding either level 2 or 3 are found throughout the Study Area.

The growth and mortality data from the bioaccumulation test with *Corbicula fluminea* performed for the tissue-residue assessment were not included in the measures of effects and exposure and LOEs (Section 3.1.4). Because these data provide a direct measure of the exposure of and effects on clams, a brief summary of the data is included. *C. fluminea* were exposed for 28 days to sediment samples collected at 33 locations in the Study Area (Figure 3-2). At the end of the exposure period the survival rates for all test sediments and negative controls ranged from 97 to 100% (a total of 177 and 183 clams were exposed in the two test set-ups). An estimate of growth rate for clams exposed to the 33 sediment samples can be made based on a comparison to the initial estimated loading biomass and the final biomass in the control sediment samples. Clams exposed to 24 of the sediment samples had similar or better growth than in the control (80% or greater of the initial estimated loading rate or the final control biomass). Clams exposed to sediment samples collected at nine locations had less growth than in the control (60 to 79% of the initial estimated loading biomass or the final control biomass). These locations included downstream and upstream of Oregon Steel Mills, Terminal 4 upstream of Slip 3, US Moorings, Gasco, Willamette Cove, Reidell Cove, Portland Shipyard, and Goldendale.

To assess the suitability of using toxicity test results for *Chironomus* and *Hyalella* as a surrogate for clams, the toxicity test results were compared with the bioaccumulation test results for laboratory clams exposed to field-collected sediment. The sediment used in the bioaccumulation testing was collected along tow lines, and the locations of these tow lines were superimposed over the *Chironomus* and *Hyalella* toxicity test sampling locations to determine where the tow lines overlapped with the sampling locations. This resulted in a comparison of results from 14 of the 33 bioaccumulation sediment sampling locations with those from 24 toxicity testing sampling locations. Of these compared sampling results, the bioaccumulation test results were in agreement with the toxicity test results 71% of the time (i.e., both tests indicated no adverse effects, or a reduction in clam growth in the bioaccumulation test concurred with pooled adverse effect endpoint [both Level 2 and Level 3] in the toxicity tests).

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The suitability of using *Chironomus* and *Hyalella* as surrogates was also evaluated using the Interspecies Correlation Estimations (ICE) software (EPA 2003b). ICE for Acute Toxicity to Aquatic Organisms and Wildlife software allows the user to predict acute toxicity to taxa with limited toxicity databases. The software was designed to address data gaps in species sensitivity and reduce reliance on UFs in ecological risk assessment. Predictions from the ICE model are based on least squares regression of existing acute toxicity data for 143 species. Relative toxicity can be predicted between species or families of interest based on similarity to the species with sufficient toxicity data. The results of the attempt at comparing relative toxicity between bivalves and amphipods were inconclusive. The ICE results did not incorporate specific life stages into the relationships overall and there were not enough data available on adult mussels (the most sediment-associated life stage) to compare general toxicity of bivalves and amphipods conclusively.

3.6.2.3 Water Assessment

The exposure of and effects on shellfish related to water were assessed by comparing near-bottom surface water and TZW data to the Eco SLs. The assessment is identical to the initial screening process for the benthic community because the same water data is used in both assessments. The Round 2 COPC exposure assessment is presented in Section 3.4.1, the Round 2 COPC effects assessment in Section 3.4.2, and the risk estimates in Section 3.6.1.3. The potential TZW iCOC exposure assessment is presented in Section 3.5.2, the potential TZW iCOC effects assessment in Section 3.5.3, and the risk estimates in Section 3.6.1.4.

3.6.3 Crayfish

Risks to crayfish were assessed based on the following LOEs: field-collected crayfish tissue data compared to aquatic TRVs, predicted tissue data compared to aquatic TRVs, and empirical surface water and TZW data compared to AWQC or TRVs.

3.6.3.1 Tissue-Residue Assessment

The exposure of and effects on crayfish were assessed by comparing tissue-residue concentrations to TRVs available in the literature. Because the tissue-residue assessment for crayfish was presented as part of risk assessment for the benthic community only a brief summary is included below. The Round 2 COPC exposure assessment is presented in Section 3.3.1, the Round 2 COPC effects assessment in Section 3.3.2, and the risk estimates in Section 3.6.1.2.1.

Copper was the only chemical identified as a potential iCOC for crayfish based on Round 1 field-collected tissue samples. No potential iCOC was identified for crayfish based on predicted tissue concentrations.

3.6.3.2 Water Assessment

The exposure of and effects on crayfish related to water were assessed by comparing near-bottom surface water and TZW data to the Eco SLs. The assessment is identical to the initial screening process for the benthic community because the same water

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data is used in both assessments. The Round 2 COPC exposure assessment is presented in Section 3.4.1, the Round 2 COPC effects assessment in Section 3.4.2, and the risk estimates in Section 3.6.1.3. The potential TZW iCOC exposure assessment is presented in Section 3.5.2, the potential TZW iCOC effects assessment in Section 3.5.3, and the risk estimates in Section 3.6.1.4.

3.7 UNCERTAINTY ASSESSMENT

This section presents the uncertainties associated with the assumptions and methods for assessing risks to the benthic community, shellfish, and crayfish, as presented in Sections 3.6.1, 3.6.2, and 3.6.3. Uncertainties are presented in the following subsections based on the predictive models (Section 3.7.1), the tissue-residue approach (Section 3.7.2), and the water assessment (Section 3.7.3).

3.7.1 Predictive Models

Several uncertainties are associated with the risk characterization of the benthic community based on the two predictive models, some related to the structure of the models, and others related to the data used for deriving the models and making predictions based on the models.

3.7.1.1 FPM Model Approach

In general, very few assumptions are made in the FPM model compared to most other existing approaches to develop SQVs, as it was designed intentionally to avoid data pre-screening or removal of outliers. No pre-processing steps are carried out, other than removal of chemicals from the dataset that have insufficient detected concentrations. This is done because the chemical distribution of each chemical used in the FPM model may affect the SQVs developed by the model. If the distribution for a given chemical has a large gap in the concentration range, a toxicity threshold is assigned to the lower concentration even though additional data might identify a threshold somewhere in the gap between the two available concentrations. Therefore, it is helpful if the gaps in the data distributions are not too large.

One underlying mathematical assumption that is not always met is that at any particular sampling location the concentration of each chemical is assumed to be independent of the concentrations of the other chemicals. In reality, there is substantial covariance between some parameters. This can be managed by grouping these chemicals into sums, conducting separate model runs for each of the covarying chemicals within a group and then combining them afterwards, or by conducting the final optimization run by hand. The approach used under the FPM for dealing with covariance depends on the degree of the effect. If a covariance analysis shows that a group of similar chemicals strongly covaries (e.g., PAHs), experience has shown that model results improve by 5 to 20% through summing them. This is done whenever possible. However, for chemicals such as metals that sometimes covary and sometimes do not, hand-optimization is used in the final model run to ensure that the metals are varied independently of one another.

For some chemical groups, such as total petroleum hydrocarbons (TPH), there is substantial uncertainty around the most appropriate method of summing them to best represent observed toxicity. TPH includes hydrocarbons of any description and many other organic compounds that could be measured analytically within the carbon range of n-C_{12} to n-C_{40}, regardless of source. To discern a relationship between sediment concentrations and toxicity is uncertain when the samples of hydrocarbon mixtures have an unknown composition of constituents with variable toxicities. The magnitude of this uncertainty is unknown because the Portland Harbor dataset did not allow exploration of some methods of summing that, from a theoretical standpoint, might be expected to improve reliability. Of the methods available, use of bulk petroleum measures such as TPH provided greater reliability (on the order of 20%) than summed PAHs or individual PAHs.

Model results are not constrained to any particular percentile of either the hit or no-hit distribution, nor required to follow a specified curve or model shape. Multiple equally valid solutions are therefore possible for some model runs, and there can be professional judgment required in selecting among them to represent the most scientifically reasonable solution (e.g., one that does not allow concentrations to fall to low levels near geologic background and is in conformance with known toxicological literature).

3.7.1.2 LRM Model Approach

The LRM approach combines predictions from the suite of individual chemical regression models into a single prediction of toxicity for the chemical mixture. Each individual chemical model uses logistic regression to describe the relationship between the probability of toxicity and the sediment concentration. Aside from the general assumptions regarding the accuracy and representativeness of the dataset (see below), several assumptions are particular to this modeling approach.

In the LRM approach, the data are pre-screened to exclude from each individual chemical model the low concentration toxic samples on the assumption that the toxicity is due to some chemical other than the one currently being modeled.

Additional data screening involves the exclusion of influential data points (i.e., high concentration non-toxic samples) on the assumption that their non-toxicity appears anomalous and on the observation that the exclusion of these data points improved the accuracy of the toxicity predictions. This step potentially reduces some of the site-specificity of the models if the excluded influential data points were from Portland Harbor.

Chemicals whose regression models fit poorly to the data (based on one or more of the following conditions: too few toxic stations retained after screening, low R^2_L value, non-significant goodness of fit) were not retained in the final suite of chemical models. Of those chemicals retained, there is no expression of the uncertainty in predictions of toxicity, or inverse predictions of threshold sediment concentrations from the logistic equations.

Finally, the individual chemical models describe the average tendency in the national dataset which may describe relationships that are not particularly relevant to Portland Harbor. For example, if antagonistic effects present in the Portland Harbor data result in reduced bioavailability of a chemical, then the Portland Harbor data may fall below the best fit regression line for that chemical. As a result, the best fit line will produce lower sediment concentration thresholds (e.g., T50 as the concentration for chemical X that is associated with a 50% probability of toxicity) than is appropriate for the Portland Harbor data. It is uncertain how the Portland Harbor data relate to the national tendency for each chemical. Only the overall predictions of toxicity that were calibrated to the Portland Harbor dataset (PrMax) should be used for toxicity predictions.

3.7.1.3 Lack of Causality

The two models are intended to predict toxicity to benthic invertebrates based on demonstrated relationships between direct measures of toxicity using the standard toxicity tests and surface sediment chemistry. However, because each sediment sample has a mixture of chemicals the models cannot establish causality. For example, the causative agents for TPH may differ among areas sampled and suggest apparent toxicity correlations that in reality are the result of a broad distribution of hydrocarbons from varying sources or elevated levels of other contaminants. Individual chemicals apparently associated with the observed toxicity may be indicators of the chemical(s) or physical conditions actually causing the toxicity.

3.7.1.4 Test Endpoints

The predicted outcome is influenced by the toxicity tests and endpoints used in the models. Different toxicity tests or endpoints ordinarily will give different predictive relationships. This was apparent with the FPM and the four different endpoints evaluated in the Benthic Interpretive Report (Windward et al. 2006). The *Chironomus* growth, *Chironomus* mortality, and *Hyalella* mortality endpoints were sensitive to similar chemicals, whereas the *Hyalella* growth endpoint showed a very different relationship. For the three endpoints with similar predicted relationships, the strongest correlation with toxicity exists for bulk hydrocarbons, PAHs, ammonia and sulfides, certain metals (e.g., cadmium, mercury, silver), and certain other organics (HCH, PCBs, DDTs, chlordane, dibutyl phthalate). The *Hyalella* growth endpoint has strong relationships only with percent fines and ammonia and has weak relationships with a limited set of metals (i.e., copper, arsenic, nickel, zinc).

3.7.1.5 Data Representativeness

The predicted outcome of the two models is based on the assumption that the chemical patterns present in the chemistry-only samples are consistent with those present in the samples used to develop the predictive relationships. If chemicals are present in the chemistry-only samples at toxic concentrations that were not present in the toxicity test samples then the models may not accurately predict toxicity (i.e., there may be unrecognized false negatives in the chemistry-only samples). If a non-toxic chemical is correlated with an unmeasured toxic chemical in the toxicity

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test samples but not in the chemistry-only samples, then the models may not accurately predict toxicity (i.e., there may be unrecognized false positives in the chemistry-only samples). The predictive ability of the models is most likely to be best when both the concentration ranges of the individual chemicals and the correlations between chemicals are similar in the toxicity test samples and in the chemistry-only samples. The chemical distribution of each chemical used in the predictive models may also affect the predictions. If the distribution for a given chemical has a gap in the concentration range, a toxicity threshold may be assigned to the lower concentration even though the "true" toxic concentration lies somewhere in the gap between the two available concentrations.

3.7.1.6 Analyte Lists

The ERA surface sediment dataset is a compilation of numerous studies performed in the Study Area. The majority of the sediment samples (932 samples) were analyzed for a suite of chemicals including metals, pesticides, SVOCs, VOCs, phenols, phthalates, PCBs, PAHs, and TPH (not all samples were analyzed for all these analytes). The toxicity prediction of the 932 samples resulted in 619 samples that were not toxic, 153 samples that were toxic, and 160 samples there were indeterminate based on the toxicity classification in Table 3-34. Because of the large list of analytes the uncertainty associated with the toxicity prediction of these samples is low. The remaining 115 sediment samples were analyzed for a limited suite of chemicals. Sixteen of these samples were only analyzed for conventionals and a few other parameters and were therefore not evaluated by either model. The FPM did not evaluate 39 samples because they were primarily analyzed for PAHs.[27] The majority of the remaining 60 samples were core samples collected by LWG. The toxicity prediction of the 46 core samples were 40 non-toxic samples, 4 toxic samples, and 2 indeterminate samples based on the toxicity classification in Table 3-34. The other 14 sediment samples were collected near the Portland Harbor Shipyard and McCormick & Baxter. Thirteen of these samples were classified as non-toxic samples and one sample as indeterminate. Because a full suite of chemicals was not associated with these samples, the toxicity prediction of these sediment samples is associated with a higher degree of uncertainty. A prediction based on a few chemicals measured may predict no toxicity, whereas if a greater number of chemicals or other chemicals had been measured, some of them may have exceeded model thresholds. Therefore, predictions at these sampling locations may underestimate risks to the benthic community based on the limited list of analytes.

3.7.1.7 Combining Model Predictions

The uncertainties discussed above affect the predictions of each modeling approach. Additional uncertainty is associated with the combined model predictions. The FPM uses three endpoints whereas the LRM uses all four endpoints including the *Hyalella* growth endpoint, the value of which has been questioned. The uncertainty of the combined model is also difficult to ascertain because of the different definitions of

[27] The FPM included TPH instead of total PAHs to predict toxicity because TPH predicted toxicity with better reliability than total or individual PAHs (Windward et al. 2006).

adverse biological affects; the FPM uses both Effect Level 2 and 3 whereas the LRM uses Effect Level 2. The predictions of the two models are being combined for policy rather than technical reasons, and as a result, the combined predictive accuracy is reduced compared to either model individually. The reason for the reduced accuracy is that each model is fully optimized using its own mathematical procedures; and by combining them, the predictive accuracy of both models is degraded.

3.7.2 Tissue-Residue Assessment

This section presents the uncertainties associated with risk characterization of benthic invertebrates using the tissue-residue approach.

Aquatic tissue-residue TRVs were derived using the following methods, as recommended by EPA: 1) calculating the fifth percentile LOAEL from all toxicological studies on aquatic organisms reported in the Ecological PRE (Windward 2005) or the fifth percentile of toxicological studies on fish reported in Dyer et al. (2000), or 2) using AWQC and BCF provided by EPA (Attachment G2). The TRVs determined which chemicals had HQs > 1.0 and therefore which chemicals were included in the initial list of Round 2 COPCs. Thus, the tendency of these TRVs to overestimate or underestimate risk and to impact the initial list of Round 2 COPCs is considered.

Using a fifth percentile LOAEL as the TRV may underestimate or overestimate risk at the individual level. LOAELs are generally appropriate at the population level and represent concentrations at which an adverse effect is observed. A NOAEL is a more protective threshold at which no effect is observed and is protective at the individual level. Tissue-residue TRVs based on the fifth percentile LOAEL also were based on all aquatic species data, including fish species. There is uncertainty in applying toxicological thresholds across all aquatic species because LOAELs that are protective of fish species may overestimate or underestimate risk to invertebrate species. Similarly, modeled TRVs based on the AWQC and BCFs may underestimate or overestimate risks to benthic invertebrates. To provide context to the aquatic TRVs, the TRVs were compared in the effect assessment to benthic invertebrate NOAELs and LOAELs available in the literature (Section 3.3.2). Table 3-44 presents a summary of the comparison for potential iCOCs.

Of the COIs identified in benthic invertebrate tissue, two were not evaluated in the Round 2 COPC screen because no data were available for the development of TRVs. No toxicological studies were identified for 2-methylnaphthalene and benzyl alcohol. The risks to invertebrate receptors associated with these chemicals for the tissue LOE is unknown because no toxicological data are available.

Several uncertainties were associated with the predicted tissue approach, as detailed in Appendix E. Site-specific BSAFs were used to predict tissue concentrations based on the best model; however, for some chemicals, there was not a strong relationship between sediment and tissue concentrations, and therefore, the “best” model was not

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a certain measure for predicting tissue concentrations. In addition, for some chemicals, it was necessary to extrapolate beyond the co-located dataset (i.e., to predict tissue concentrations at higher sediment concentrations than were measured in the co-located subset of the data). There is high uncertainty in applying the BSAFs to concentrations outside the range of empirical concentrations used to develop the model

Some of the BSAFs were uncertain because they were based on one or a few co-located data pairs because of low detection frequencies. These BSAFs are more highly uncertain than the BSAFs developed for more frequently detected chemicals. High uncertainty is associated with the BSAF models developed for beta-HCH and endrin. For both of these chemicals, the majority of the higher concentrations were either NJ-qualified or non-detects (included in the models as one-half the DL).

3.7.3 Surface Water Assessment

This section presents the uncertainties associated with risk characterization of benthic invertebrates using the near bottom surface water data. The uncertainties include selection of Eco SLs, surface water sampling methods, flow regimen during the sampling events, risk assessment of total lead, and the use of NJ-qualified data at a location downstream of Oregon Steel Mills (W01).

The appropriateness of the Eco SLs was evaluated as the primary uncertainty associated with the effects assessment for the surface water LOE. Eco SLs are based AWQC or ODEQ benchmarks that were developed based on fish or invertebrate species sensitivities and are considered protective of all aquatic receptors, including benthic invertebrates. The AWQCs for 4,4′-DDT were used as the Eco SLs for all DDD and DDT isomers, and total DDTs. A large portion of the data used to develop the acute AWQC was invertebrate studies. The chronic AWQC was derived using the geometric mean of normalized BCFs and a lipid value of 8%. Uncertainties are associated with the derivation of the chronic AWQC which may under or overestimate risks from exposure to 4,4′-DDT. Similarly, using the 4,4′-DDT AWQCs to assess risks from all DDD and DDT isomers, and total DDTs may under or overestimate risks from these chemicals. The Eco SL for 4-chloro-3-methyl phenol has some uncertainty associated with its use in assessing risks to fish. The chronic Eco SL for 4-chloro-3-methylphenol was calculated from an acute ODEQ value using a UF of 50, as directed by EPA (2006d). This chronic Eco SL may overestimate toxicity to fish receptors.

The surface water samples were collected by two methods, peristaltic pump and XAD. All of the Round 2 COPCs, except 2,4′-DDD, 4,4′-DDD, and 4,4′-DDT, were identified in samples collected with the peristaltic pump. The peristaltic pump collects a water sample in relatively short time and the sample represents a snapshot of the exposure to benthic invertebrates that may over or underestimate risks. The XAD collects a water sample over numerous hours (approximately 13 hours at a rate of

1.25 L per minute) and may therefore overestimate the exposure concentrations to benthic invertebrates at any given time.

The surface sampling events were planned to collect water samples at high- and low-flow conditions and in the spring to assess the exposure of amphibians. However, because the November sampling event did not occur under high-flow conditions, this exposure scenario could not be addressed in this report. Additional data are currently being collected in Round 3.

Total lead was identified as a Round 2 COPC based on exceedances of the chronic Eco SL (max HQ of 3.3) in four near bottom water samples collected at the head of the International Slip (W004) and Terminal 4 Slip 3 (W008). The Eco SLs for lead are based on dissolved lead. The chronic Eco SL for total lead is calculated as the acute Eco SL for dissolved lead divided by a factor of 50. Total lead includes lead that is occluded in minerals, clays, and sand or is strongly sorbed to particulate matter and is not toxic or likely to become toxic under natural conditions whereas dissolved lead measures all forms of lead that are toxic to aquatic organisms or can readily be converted to toxic forms under natural conditions. Risk characterization to benthic invertebrates based on a comparison to the chronic Eco SL is therefore highly uncertainty and overestimates the risks associated with total lead.

HQs for 2,4′-DDT and total DDTs downstream of Oregon Steel Mills (W001) are based on a single NJ-qualified result. 2,4′-DDT is the only isomer that was detected in this sample. NJ-qualification indicates "the presence of an analyte that has been 'tentatively identified,' and the associated numerical value represents its approximate concentration" (EPA 1999). The qualification indicates that the analyst believed that the result was due to analytical interference from a chemical other than the target analyte. The NJ-qualified result is biased high and will result in an overestimation of risk.

3.7.4 Transition Zone Water Assessment

This section presents the uncertainties associated with the assumptions and methods used to identify TZW iCOCs. Uncertainties associated with the TZW framework involved sampling methods, using TZW data for exposure estimates, sampling locations, NJ-qualified data, and COIs without Eco SLs.

The TZW samples were collected with Trident and peeper (both filtered and unfiltered). Uncertainty is associated with whether these samples represent appropriate estimations of TZW exposure to the benthic community. Additional uncertainty is associated with whether the TZW data represent appropriate exposure for the benthic community. TZW ventilation for benthic fish and invertebrates is often limited because of low oxygen concentrations and low food content in TZW (Arnot and Gobas 2004). Therefore, the exposure of benthic fish and invertebrates to TZW may be overestimated if these organisms are ventilating near-bottom surface

water, rather than TZW. Based on this uncertainty, a TZW ventilation assumption of 1 to 10% was used for benthic fish and invertebrates.

The 10 TZW sampling locations were selected because they have a confirmed or reasonable likelihood for discharge of upland COIs to Portland Harbor (Integral 2006). Samples taken at these sites may overestimate site-wide risk of benthic fish and invertebrates to TZW.

HQs for 2,4′-DDT and total DDTs at Rhone Poulenc are based on an NJ-qualified results. 2,4′-DDT is the only isomer that was detected in this sample. NJ-qualification indicates "the presence of an analyte that has been 'tentatively identified,' and the associated numerical value represents its approximate concentration" (EPA 1999). The qualification indicates that the analyst believed that the result was due to analytical interference from a chemical other than the target analyte. The NJ-qualified result is biased high and will result in an overestimation of risk.

Some COIs in TZW were not evaluated in the Round 2 COPC screen because no data were available for the development of Eco SLs. No toxicological studies were identified for calcium, titanium, diesel-range hydrocarbons, gasoline-range hydrocarbons, residual-range hydrocarbons, TPH, and individual dioxins and furans. The risks to invertebrate receptors associated with these chemicals for the TZW LOE is unknown because no toxicological data are available.

3.8 RISK CONCLUSIONS

This section presents the risk conclusions including iCOCs identified for the benthic community, shellfish, and crayfish. The risk conclusions for the benthic community based on toxicity testing and the FPM are presented in Section 3.8.1, and the risk conclusions for all benthic invertebrate receptors based on all other LOEs are presented in Section 3.8.2.

3.8.1 Sediment Toxicity Testing and FPM

The FPM, in combination with toxicity testing, was identified in Section 3.6.1.1 as providing the best prediction of risks to the benthic community. Additional discussion of uncertainties associated with the modeling effort was provided in Section 3.7.1. Because the FPM cannot establish causality between chemicals and toxicity, the chemicals associated with toxicity based on the FPM are identified as potential iCOCs for this Round 2 ERA. Final COCs will be identified as part of the BERA. The results of the toxicity testing and predictions of toxicity based on the FPM are presented in Figure 3-8, and potential iCOCs are presented in Table 3-35. Further assessment of the toxicity testing results and FPM predictions was conducted using cluster analysis (presented in Section 10 of the Comprehensive Round 2 Report) to identify iAOPCs for the benthic community. Potential data needs for the benthic community in the BERA are presented in Section 8.0.

3.8.2 Tissue-Residue, Surface Water, and TZW LOEs

The HQs presented in Sections 3.6.1.2, 3.6.1.3, and 3.6.1.4 were combined with the uncertainties associated with each Round 2 COPC/receptor risk approach presented in Section 3.7.2, 3.7.3, and 3.7.4 to arrive at risk concentrations. Final COCs will be identified as part of the BERA; selected iCOCs have been identified for this Round 2 ERA based on the following rationale:

- Empirical and predicted tissue Round 2 COPCs that exceeded the aquatic TRVs[28] based on the UCL EPC were identified as iCOCs, except for COPCs with single outlier data points, NJ-qualified data, or non-detects causing the UCL HQ exceedance.
- Near-bottom surface water Round 2 COPCs that exceeded the Eco SLs based on the UCL EPC were identified as potential iCOCs. All potential iCOCs were included as iCOC for benthic invertebrates, if supported by other LOEs.
- Potential TZW iCOCs were identified through the TZW framework. All potential iCOCs were included as iCOC for benthic invertebrates, if supported by other LOEs.

Based on the above rationale, the following chemicals were identified as iCOCs for benthic invertebrates:

Cadmium – Cadmium was identified as an iCOC for the benthic community and shellfish based on the tissue LOE. A site-wide UCL HQ of 1.2 for cadmium was calculated based on field-collected clam tissue samples. A predicted site-wide UCL HQ of 12 was calculated based on field-collected clams. The UCL HQs for crayfish and laboratory-exposed clams and worms were < 1.0. Analysis of near-bottom surface water and TZW LOEs did not identify cadmium as a potential iCOC.

Copper – Copper was identified as an iCOC for the benthic community, shellfish, and crayfish based on the tissue LOE. The site-wide UCL HQs for both field-collected tissue (clams and crayfish) and laboratory-exposed tissue (clams and worms) samples ranged from 1.2 to 4.8. A predicted site-wide HQ of 1.3 was calculated based on laboratory-exposed clams. Analysis of near-bottom surface water and TZW LOEs did not identify copper as a potential iCOC.

Zinc – Zinc was identified as an iCOC for the benthic community and shellfish based on the tissue LOE. A site-wide UCL HQ of 1.4 for zinc was calculated based on field-collected clam tissue samples. A predicted site-wide UCL HQ of 1.3 was calculated based on field-collected clams. The UCL HQs for crayfish and laboratory-exposed clams and worms were < 1.0. Analysis of near-bottom surface water and TZW LOEs did not identify zinc as a potential iCOC.

[28] For TBT, the TRV based on Meador et al. (2002) was used in the assessment.

PAHs – Total PAHs and three individual PAHs (i.e., benzo(a)pyrene, chrysene, and pyrene) were identified as iCOCs for the benthic community, shellfish, and crayfish based on the tissue, near-bottom surface water, and TZW LOEs. A predicted site-wide UCL HQ of 4.2 for benzo(a)pyrene and a site-wide UCL HQ of 1.7 for chrysene were calculated based on laboratory worm tissue samples. Site-wide UCL HQs of 4.6, 1.4, and 3.8 for pyrene were calculated for laboratory-exposed worms and predicted tissue concentrations of laboratory-exposed clams and worms, respectively. A site-wide UCL HQ of 16 for total PAHs was calculated based on laboratory-exposed worm tissue samples; and predicted site-wide UCL HQs of 2.8, 4.1, and 9.8 were calculated for total PAHs based on field-collected clams, laboratory-exposed clams, and laboratory-exposed worms, respectively. The UCL HQs for crayfish and laboratory-exposed and field-collected clams were < 1.0. A site-wide UCL HQ of 1.1 for benzo(a)pyrene was calculated based on near-bottom surface water samples. The maximum HQs for the three individual PAHs based on the TZW framework were 2,700 for benzo(a)pyrene, 17 for chrysene, and 15 for pyrene.

PCBs – Total PCBs were identified as an iCOC for the benthic community based on the tissue LOE. A site-wide UCL HQ of 3.3 for total PCBs was calculated based on laboratory worm tissue samples. The UCL HQs for crayfish and laboratory-exposed and field-collected clams were < 1.0. Analysis of near-bottom surface water LOE resulted in HQs < 1.0. PCBs were not analyzed in TZW.

DDTs – DDTs (including 2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDE, 4,4′-DDT, and total DDTs) were identified as iCOCs for the benthic community, shellfish, and crayfish based on the tissue, near-bottom surface water, and TZW LOEs. Site-wide UCL HQs for 4,4′-DDD in field-collected clam tissue samples and laboratory-exposed clam and worm tissue samples ranged from 1.5 to 7.8. Site-wide UCL HQs of 1.1 and 1.2 were calculated for total DDTs based on laboratory-exposed worm and clam tissue samples. Site-wide UCL HQs ranging from 2.1 to 10 were calculated for 2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDT and total DDTs based on near-bottom surface water samples. In addition, 2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDE, 4,4′-DDT, and total DDTs were identified as iCOCs based on the TZW framework.

Cyanide, perchlorate, 13 individual PAHs (i.e., 2-methylnaphthalene, acenaphthene, anthracene, benzo[a]anthracene, benzo[b]fluoranthene, benzo[g,h,i]perylene, benzo[k]fluoranthene, dibenzo[a,h]anthracene, fluoranthene, fluorene, indeno[1,2,3-cd]pyrene, naphthalene, and phenanthrene), TBT, BEHP, and dibutyl phthalate were retained as potential iCOCs for benthic invertebrates. Potential iCOCs for benthic invertebrates were those chemicals associated with high uncertainty or identified by the TZW LOE, but not supported by other LOEs, as posing risks to benthic invertebrates. These potential iCOCs are not expected to pose significant risks to benthic invertebrates because there is limited potential for exposure.

No other Round 2 COPCs were identified as iCOCs or recommended to be carried forward to the BERA based on the tissue-residue, near-bottom surface water, or TZW

LOEs. The iCOCs identified through the tissue-residue, near-bottom surface water, and TZW LOEs were not used to identify iAOPCs for benthic invertebrate, instead iAOPCs were identified by the toxicity testing results and the FPM (Section 10 of the Comprehensive Round 2 Report). Potential data needs for benthic invertebrates in the BERA are presented in Section 8.0.

4.0 FISH RISK ASSESSMENT

This section presents the Round 2 ERA for fish in the Portland Harbor Study Area. Section 4.1 identifies the selected ecological fish receptors, exposure pathways, and LOEs evaluated for determining risks to fish species that use the LWR and the results of the screening assessment to identify fish Round 2 COPCs. Round 2 COPCs for each line of evidence were identified using data collected to date based on screening level evaluations for each line of evidence. Sections 4.2 though 4.5 present the exposure and effects methods for evaluating risks to fish receptors via each line of evidence for assessing risks for all Round 2 COPCs. These LOEs include the critical tissue-residue assessment (Section 4.2), dietary assessment (Section 4.3), surface water assessment (Section 4.4), and the TZW assessment (Section 4.5). Round 2 COPCs are carried forward in a risk evaluation to identify the iCOCs that will be further evaluated in the BERA. Section 4.6 presents the Round 2 risk conclusions and iCOCs that will inform the BERA.

4.1 PROBLEM FORMULATION

This section summarizes the problem formulation for fish receptors in the LWR. Details on the fish problem formulation are presented in Appendix B of the Programmatic Work Plan (Integral et al. 2004b).

4.1.1 Fish Resources Potentially at Risk

Numerous fish species use the Willamette River. Fish species that were identified as using habitat within the Study Area were grouped into the following feeding guilds:

- **Omnivores/herbivores** – fish that feed on vegetation or vegetation and invertebrates
- **Benthopelagic/benthic invertivores**– fish that feed primarily on invertebrates living either in the water column or on bottom substrates
- **Piscivores** –fish that feed primarily on fish
- **Detritivores** – fish that feed primarily on detritus

Appendix B of the Programmatic Work Plan (Integral et al. 2004b) provides detailed information on prey and habitat preferences and Willamette River site use of fish species within these feeding guilds. Table 4-1 presents the fish species known to be present or common in the LWR. Of the fish species with a confirmed presence in the LWR, seven (i.e., green sturgeon, coastal cutthroat trout, steelhead, chinook salmon, coho salmon, Pacific lamprey, and river lamprey) are identified as special-status

species.[29] Table 4-2 presents a list of fish species with rare or possible but unconfirmed presence in the LWR. Three fish species (i.e., chum salmon, Oregon chub, and bull trout) that were identified as species that have a rare or unconfirmed presence in the LWR were also identified as special-status species.

4.1.2 Selected Receptors of Concern

To select representative fish species, fish in the Study Area were evaluated using the systematic criteria presented in Appendix B of the Programmatic Work Plan (Integral et al. 2004b) and divided into primary feeding guilds as identified in Section 4.1.1 (i.e., omnivorous and herbivorous fish, invertivorous fish, piscivorous fish, and detritivorous fish). Invertivorous fish were further divided into benthic feeders, and salmonid and non-salmonid benthopelagic feeders. The following fish species were selected as ecological receptors to represent the various feeding guilds in the LWR.

- **Omnivorous and herbivorous fish** – largescale sucker, carp, and pre-breeding white sturgeon[30]
- **Invertivorous fish** – sculpin (benthic invertivore) and peamouth and juvenile chinook salmon (benthopelagic invertivores)
- **Piscivorous fish** – smallmouth bass and northern pikeminnow
- **Detritivorous fish** – Pacific lamprey ammocoetes

These selected receptor groups were presented in Appendix B of the Programmatic Work Plan (Integral et al. 2004b), which was conditionally approved by EPA and its partners in June 2004. The following subsections briefly summarize the rationale for selecting the ecological receptor groups that will be assessed for the Portland Harbor Superfund Study Area.

4.1.2.1 Largescale Sucker, Carp, and White Sturgeon

Omnivorous and herbivorous fish in the LWR are exposed to chemicals primarily through their diet and incidental ingestion of sediment and water. Omnivores are predominately bottom feeders. The common omnivores in the Study Area are largescale sucker, carp, and white sturgeon. These species are benthic feeders that ingest sediment along with a variety of animal, plant, and detrital material. All of these species are long-lived, with life expectancies of 15, 20, and 100 years for largescale sucker, carp, and white sturgeon, respectively.

Largescale sucker was selected as a receptor of concern over carp to represent omnivorous and herbivorous fish in the Study Area because of their close association with sediments and because they are one of the most abundant species in the LWR. Use of this species as a surrogate for herbivores was considered protective because

[29] Special-status species include federal and state proposed and candidate species.
[30] These three omnivores were used to represent herbivores due to their potential for greater chemical exposure and low sensitivity of herbivores present in the LWR to chemical exposure.

largescale suckers have a higher trophic status than the two herbivorous fish (chiselmouth and mountain sucker) that occur in the LWR and thus potentially have a higher exposure to biomagnifying chemicals. In addition, herbivorous fish are not known to be sensitive to environmental pollutants. Carp was included as a surrogate receptor of concern for dioxin-like chemicals, including PCB congeners, per EPA request.

As stated in Appendix B of the Programmatic Work Plan (Integral et al. 2004b), white sturgeon also was selected as a receptor of concern to represent omnivorous and herbivorous fish in the Study Area per EPA's request. Some uncertainty exists in the selection of white sturgeon as an ecological receptor because this species has a home range that extends beyond the Study Area. Juvenile (pre-breeding) sturgeon were assessed in this Round 2 ERA and were assumed to be protective of adult sturgeon because the site use of pre-breeding sturgeon is expected to be higher than that of adult sturgeon.

4.1.2.2 Juvenile Chinook Salmon, Sculpin, and Peamouth

Many species of invertivorous fish inhabit the LWR. Carnivorous fish are potentially exposed to chemicals through their prey and through direct and indirect exposure to water and sediments. Within the invertivorous feeding guild, two resident fish species were selected, along with an anadromous salmonid species. Sculpin was selected to represent invertivorous benthic fish because of their close association with sediments and their small home range. As resident invertivores, sculpin are exposed to chemicals in the Study Area during all life stages. Peamouth was also selected to represent invertivorous fish because they are a resident species with a home range that is potentially much larger than that of sculpin. Peamouth feed on both benthic and pelagic prey, whereas sculpin feed on benthic prey.

Juvenile chinook salmon was also selected to represent all juvenile salmonids in the Study Area. Juvenile salmonid residence time in the LWR and within the Study Area is a matter of ongoing investigation, and there is uncertainty in the year-to-year variation of size and age cohorts among the various species. However, of the juvenile Pacific salmonids, juvenile chinook salmon appear to have the longest residence in the Study Area(North et al. 2002; Ward and Nigro 1992), giving them the greatest potential exposure to chemicals. Given the uncertainty regarding resident times, peamouth (which have similar habitat and feeding habits to juvenile salmonids, but are year-round residents), was used to conservatively estimate an upper level of exposure for juvenile chinook.

Both juvenile and adult Pacific salmon species, including chinook salmon, migrate through the Study Area; however, during the adult life stage, these salmonids are exposed to chemicals in their oceanic habitat. Thus, the source of chemical burdens in adult fish found in the site would be difficult to ascertain. During their upstream migrations, most anadromous species in the adult life stage feed infrequently, so exposure to chemicals in the Study Area through the dietary pathway is likely to be

relatively low. These factors make adult salmonids an inappropriate choice for a representative species. In addition, juveniles are generally believed to be more susceptible to toxic substances than adults.

4.1.2.3 Smallmouth Bass and Northern Pikeminnow

Northern pikeminnow, smallmouth bass, largemouth bass, black crappie, white crappie, and walleye are common piscivores in the LWR. As high trophic level predators, all of these fish play a key role in the dynamics of the aquatic community. Because of their high trophic status, these fish have a greater potential for biomagnifying chemicals such as PCBs, DDTs, and mercury than many other species. The diets of piscivorous fish in the LWR have been shown to be similar (Fishman 1999). Of the piscivores listed in Table 4-1, northern pikeminnow is the only native species.

Northern pikeminnow was selected as representative of piscivorous fish because it is relatively long-lived and is a top predator. Northern pikeminnow, black crappie, and white crappie are more common in the LWR than the other piscivorous fish. Smallmouth bass were also selected as representative of piscivorous fish because of their smaller home range and similarity to other common centrarchid piscivores in the LWR.

4.1.2.4 Pacific Lamprey Ammocoetes

Pacific lamprey ammocoetes were selected as the detritivorous fish receptor in the LWR. Lamprey ammocoetes are unique to the fish community in the Study Area because they live burrowed in the sediment and filter algae, detritus, and other organic material from the near-bottom water column. These are the only detritivorous fish present in the LWR. Lamprey ammocoetes reside in freshwater for up to 6 years and potentially have exposure to chemicals in sediment. Pacific lamprey are a species of concern under the Endangered Species Act (ESA) and an Oregon State sensitive species. Pacific lamprey are valued by Native American tribes in the region for ceremonial purposes. Adult lamprey travel through the Study Area while migrating to upstream spawning areas in the upper river, and the amount of time that they spend in the Study Area is unknown. Furthermore, adult lamprey do not feed in freshwater; their growth results primarily from parasitic feeding on other fish in the ocean or estuary. Therefore, their exposure through the dietary pathway while migrating through the Study Area is insignificant. Due to the differential exposure of the life stages, ammocoetes are a more appropriate life stage from which to assess risks to this species and/or feeding guild, and adult lamprey were not proposed as an ecological receptor of concern.

4.1.3 Exposure Pathways

Potential chemical exposure pathways for the selected fish receptors in Portland Harbor were identified in Appendix B of the Programmatic Work Plan (Integral et al. 2004b). Receptors may be exposed to chemicals in water or sediment, either through direct (i.e., dermal) contact with or ingestion of sediments or water or indirectly

through the ingestion of food. The CSM was developed for the Programmatic Work Plan and is presented in Figure 4-1. The CSM illustrates the pathways that chemicals may follow from primary sources to the ecological receptors.

The ingestion of prey (i.e., biota) and incidental ingestion of sediment were considered complete pathways for all fish receptors. All fish are in direct contact with surface water; this pathway was considered to be complete and major and thus evaluated quantitatively in the surface water line of evidence. Direct contact with sediment was considered a complete and major pathways for benthic fish, including largescale sucker, carp, white sturgeon, sculpin, and lamprey ammocoetes.

Incidental ingestion of surface water may occur for all fish species during feeding, but although this pathway is considered complete, because freshwater fish do not drink water it is minor when compared to other exposure routes. In addition, TZW exposure by direct contact (while foraging for prey) and ingestion are considered incomplete pathways for all fish receptors. However, TZW exposure for sculpin and lamprey ammocoetes was evaluated per direction from EPA, and the evaluation is presented in Section 4.5.

4.1.4 Lines of Evidence Approach and Methods

LOEs (pairings of individual receptors and exposure pathways or effects) for assessing risks to fish were presented in Appendix B of the Programmatic Work Plan (Integral et al. 2004c) and were further refined based on discussions between LWG, EPA, and EPA's partners (EPA 2004a, 2005a). LWG agreed to evaluate all of the LOEs for assessing risks to fish receptors (EPA 2006a) (Table 4-3).

The weighting scheme for the LOEs for fish depended on the purpose of the assessment/task, of which there were four:

- Screening COIs to identify Round 2 COPCs
- Further screening of Round 2 COPCs to identify iCOCs
- Identifying potential data needs for the BERA
- Providing information that would be used to identify iAOPCs

Round 2 COPCs were identified in the SLERA. The other three tasks were done to better prepare for the BERA and other parts of the RI/FS that may use risk assessment results.

The weight-of-evidence approach for Round 2 COPC identification was as follows:

- Round 2 COPCs were identified for each LOE.
- The Round 2 COPC lists were integrated across LOEs to get the overall list of Round 2 COPCs for fish.

This is a conservative approach in that any chemical that screened in based on any one LOE was carried forward as a Round 2 COPC.

The identification of iCOC, potential data needs, and iAOPCs used a weight-of-evidence approach similar to that used for Round 2 COPC identification, differing in level of conservatism. The identification of Round 2 COPCs used an appropriate level of conservatism for the SLERA. The iCOC identification used less-conservative assumptions and incorporated a weighted approach across LOEs in an effort to narrow the list of chemicals and identify the potential areas associated with risks to fish in the Study Area. All LOEs for were used to identify iCOCs, with the exception of the salmonid olfactory function LOE and the lesion occurrence in benthic fish LOE, which were given a weight of zero, because no risks to fish from these LOEs were determine in the SLERA.

These LOEs are evaluated in the following sections. Section 4.2 presents the approach and results of the critical tissue-residue LOE using empirical and predicted tissue data. Section 4.3 presents the LOE based on dietary doses; Section 4.4 presents the LOE based on surface water exposure. A separate discussion of the potential effects of dissolved metals (specifically copper) on salmonid olfactory function is provided in Attachment G4. Although TZW exposure is expected to be an incomplete pathway of exposure for fish receptors, the TZW analysis for sculpin and lamprey ammocoetes is presented in Section 4.5, per EPA request. Attachment G4 presents an evaluation sediment chemistry data relative to sediment quality guidelines derived from consideration of the occurrence of skin and liver lesions associated with direct exposure of benthic fish to PAHs in sediment.

4.1.5 Screening Summary for Round 2 COPC Identification

Based on the R2 COIs identified in Section 2.2,a screening step was conducted for each LOE presented in Table 4-3 to identify Round 2 COPCs for the fish risk evaluation. Attachment G4 presents details on the screening process and identification of Round 2 COPCs for the tissue-residue, dietary dose, and water LOEs, and a summary of the results are presented in the following subsections. Attachment G4 also presents an evaluation of additional LOEs (impaired chinook olfactory function associated with surface water exposure and lesion occurrence associated with direct exposure to PAHs in sediment). Data used in the COPC screens are described in Section 2.

4.1.5.1 Tissue-Residue Screen and Round 2 COPC Identification

To identify Round 2 COPCs for fish using the critical tissue-residue approach, the maximum whole-body tissue concentration[31] of each COI was compared to an SL TRV for that chemical. If the maximum exposure concentration (for each receptor) was greater than the SL TRV, the chemical was identified as a Round 2 COPC. The

[31] Maximum concentrations were defined as the higher of the maximum detected concentration or the maximum reporting limit (RL) of a non-detected concentration

identification of Round 2 COPCs by means of a conservative screen in which maximum concentrations in relevant media are used is consistent with EPA guidance (EPA 1997a, b) and an important step for narrowing the list of chemicals to only those chemicals that could potentially pose a risk to selected fish receptors.

Because sculpin are associated with small exposure areas, a predictive tissue-residue screen was run for sculpin to ensure that the empirical tissue data weren't under-representing an individual exposed to the 95th percentile site-wide sediment concentration. This predicted tissue screen used site-specific biota-sediment accumulation relationships to estimate what the tissue concentration might be for a sculpin exposed to the 95th percentile site-wide sediment concentration. The predicted tissue screen identified two Round 2 COPCs, one of which (BEHP) was previously identified as a Round 2 COPC based on the empirical tissue-residue screen. The additional R2 COPC identified by the predicted tissue screen was selenium.

Based on the results of the tissue-residue screening, 10 COPCs were identified for fish receptors: chromium, lead, mercury, zinc, BEHP, total PCBs, 4,4′-DDD, 4,4′-DDT, total DDTs, and beta-HCH.A summary of the fish tissue Round 2 COPCs identified for each fish receptor is presented in Table 4-4.

4.1.5.2 Dietary Dose Screen and Round 2 COPC Identification

To identify dietary Round 2 COPCs for fish, the maximum estimated exposure dose (IR_{diet}) of each dietary fish COI was compared to a NOAEL TRV for that chemical. If the maximum exposure dose (for each receptor) was greater than the NOAEL TRV, the chemical was identified as a Round 2 COPC. The identification of Round 2 COPCs through a conservative screen in which maximum concentrations in relevant media are used is consistent with EPA guidance (EPA 1997a, b) and an important step for narrowing the list of chemicals to only those that could potentially pose a risk to ecological receptors.

Based on the results of the dietary dose screening, eight COPCs were identified for fish receptors: cadmium, copper, mercury, TBT, benzo(a)pyrene, total PAHs, total PCBs, and total DDTs. A summary of the fish dietary dose Round 2 COPCs that were evaluated for each fish receptor is presented in Table 4-5.

4.1.5.3 Water Screen and Round 2 COPC Identification

Attachment G4 presents the screening process and identification of Round 2 COPCs for the surface water and TZW LOEs.

4.1.5.3.1 Surface Water

To identify Round 2 COPCs for fish, maximum detected COI concentrations in surface water were compared to the ecological surface water screening level (Eco SL) for that chemical. If the maximum concentration was greater than the Eco SL, the chemical was identified as a Round 2 COPC.

Based on the results of the surface water screening presented in Attachment G4, 10 COIs had maximum surface water concentrations greater than chronic Eco SLs (zinc, benzo(a)anthracene, 4-chloro-3-methyl phenol, total PCBs, 2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDT, total DDTs). A summary of the surface water Round 2 COPCs is presented in Table 4-6.

4.1.5.3.2 Transition Zone Water

The identification of TZW Round 2 COPCs for fish receptors is the same as that for benthic invertebrates (Section 3.0), in which exposure was evaluated by comparing maximum detected chemical concentrations in TZW to Eco SLs. Based on the results of the TZW screening, 55 COPCs were identified, as presented in Table 4-7.

4.2 CRITICAL TISSUE-RESIDUE ASSESSMENT

The critical tissue-residue assessment was one LOE for evaluating risks to selected fish receptors. In this assessment, whole-body tissue concentrations were compared to tissue TRVs for each chemical and fish receptor pair. This LOE incorporated all of the exposure pathways for fish. All Round 2 COPC/receptor pairs identified for the tissue-residue LOE in Table 4-4 (see Section 4.1.5) were evaluated for the tissue assessment LOE and are presented in the following section.

4.2.1 Round 2 COPC Exposure Assessment

The whole-body tissue concentration integrates exposure from all pathways (i.e., direct contact with sediment and water, ingestion of prey, and incidental ingestion of water and sediment). The tissue EPCs of all the Round 2 COPC/receptor pairs identified in Table 4-4 are presented in the following section.

Tissue EPCs for fish were represented by: 1) the individual composite sample concentrations; and 2) the UCL on the mean concentration over all composites. Figures 4-2 through 4-8 present the locations of fish collected for each fish composite sample. EPA's comments on the Ecological PRE state that "Evaluation of tissue residues for the purposes of evaluating fish health should not include a mean value calculated from composite means, especially of composites that were taken over an area larger than a localized population. Although this may provide information about the range of risk within the Study Area, future evaluations should compare each composite individually against appropriate TRVs, and the areas exceeding reported" (EPA 2006b). LWG evaluated tissue EPCs for fish as individual composite samples in response to this comment. However, LWG also evaluated EPCs as the UCL of the mean composite sample concentration.

Two Round 2 COPCs (i.e., selenium and BEHP) were identified for sculpin through the predicted tissue screening process presented in Attachment G4. The predicted

site-wide sculpin tissue concentration for selenium and BEHP based on the site-wide UCL sediment concentration were 1.73 mg/kg and 1,690 mg/kg-lipid,[32] respectively.

UCL tissue EPCs were calculated for all Round 2 COPCs using ProUCL v. 3.0 software (EPA 2004d), which was developed by EPA to compute an appropriate 95th UCL of an unknown population mean. ProUCL tests for normality, lognormality, and a gamma distribution of the dataset and computes a conservative and stable UCL of the unknown population mean (EPA 2004c). Prior to running data through ProUCL to determine UCLs, non-detected data were adjusted to one-half the detection limit to represent the concentration value.[33] For each Round 2 COPC, the UCL recommended by ProUCL was used as the EPC for the risk calculations. In the case where an insufficient unique number of data values was available, the maximum concentration[34] was used to represent the EPC. Attachment G1 presents the summary statistics (i.e., minimum, maximum, and mean Round 2 COPC concentrations), distribution types, ProUCL-recommended UCLs, and selected EPCs for each Round 2 COPC. EPC tissue concentrations for each Round 2 COPC/receptor pair are presented in Tables 4-8 through 4-13.

4.2.2 Round 2 COPC Effects Assessment

This section presents a brief summary of the toxicity literature for each of the fish tissue Round 2 COPCs. The approach for identifying fish SL TRVs was developed by EPA and its partners for data evaluation in the Ecological PRE (EPA 2005b; Windward 2005). In this conservative approach, one TRV was developed for each chemical to be protective of all aquatic species (i.e., representative of all aquatic receptors, both fish and invertebrates), rather than multiple TRVs (i.e., NOAELs and LOAELs) for each receptor group. These SL TRVs were used to identify fish tissue Round 2 COPCs, as presented in Attachment G4.

Prior to the developing an approach for selecting SL TRVs, an extensive search for available toxicological literature on fish tissue COIs was conducted to derive NOAEL and LOAEL TRVs for fish. Unlike the SL TRVs, which are based on aquatic organisms (both fish and invertebrate species), the NOAEL and LOAEL TRVs are based on fish-specific toxicity data. Further, the fish-specific TRVs are derived from studies that relate to the endpoints being evaluated for fish receptors (i.e., survival, growth, and reproduction) and were extensively reviewed for acceptability (e.g., whether test conditions were controlled) and their ecological relevance to the exposure conditions of fish receptors in the Study Area. The fish TRV literature search is described in detail in Appendix B of the Ecological PRE (Windward 2005).

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[32] The average sculpin lipid value, 4.17%, was used to estimate lipid-normalized BEHP tissue concentrations in sculpin.
[33] Use of a regression on order statistic to estimate values to represent non-detected concentrations may be appropriate for calculation of UCLs in the BERA.
[34] Where the maximum concentration was a non-detected value, the full detect limit was used to represent the EPC.

The Round 2 COPC risk analysis was based on the LWG-recommended NOAEL and LOAEL TRVs because the use of SL TRVs is uncertain for evaluating risks to fish (e.g., SL TRVs are not fish-specific, do not exclude unacceptable studies, and do not differentiate between effect and no-effect threshold levels). The TRVs selected for use in this Round 2 report are consistent with the TRVs recommended in Appendix B of the Ecological PRE (Windward 2005), with the exception that selected Round 2 TRVs were not based on studies that measured effects in field-collected organisms, had a value based on behavior endpoints, or reported tissue concentrations measured in individuals poorly representative of those in which adverse effects were observed.

Injection and behavior studies were evaluated as part of the TRV review process, per EPA's direction (June 30, 2004 comments, EPA 2004b). However, for the Round 2 ERA, behavior studies were not considered direct measures of the survival, growth, and mortality endpoints that were selected and agreed upon for this fish ERA (Table 4-3). Toxicity studies based on field-collected organisms are not appropriate for evaluating risks beyond a screening-level evaluation because the presence of multiple chemicals and other environmental factors complicates the interpretation of field study results. Studies that reported tissue concentrations that were not measured at the time that an adverse effect was observed do not represent critical tissue thresholds. The TRV technical memorandum addendum (Attachment G5) presents the changes in recommended NOAEL and LOAEL TRVs for this Round 2 ERA. For all other Round 2 COPCs, selected NOAEL and LOAEL TRVs are based on those recommended in Appendix B of the Ecological PRE (Windward 2005)

Summaries of the toxicological studies and selected NOAEL and LOAEL TRVs for each fish tissue Round 2 COPC are presented in the following subsections. Toxicological data presented in this section are assessed in combination with exposure data (Section 4.2.3) in the risk characterization (Section 4.6.1).

4.2.2.1 Chromium

Only one toxicity study that measured effects and whole-body chromium concentrations associated with chromium exposure was evaluated for TRV selection (Table 4-14). Fingerling rainbow trout were exposed to various concentrations of aqueous hexavalent chromium at two pH levels (7.8 and 6.5) for up to four days (van der Putte et al. 1981) Fifty percent mortality was observed in fish exposed to chromium at pH 7.8 and with tissue residues of 10.5 mg/kg ww. Twenty-five percent mortality was observed in fish exposed to chromium at pH 6.5 and with tissue residues of 8.7 mg/kg ww. The lower LOAEL (based on the two pH levels) of 8.7 mg/kg ww was selected as the LOAEL TRV for chromium. No effect on rainbow trout survival was observed in fish exposed to 2.0 and 5.0 mg/L chromium at pH 6.5 and 7.8, respectively. The range of tissue concentrations where no mortality was reported was 0.9 to 5.5 mg/kg ww. The upper range where no effects were observed, 5.5 mg/kg ww, was selected as the NOAEL TRV for chromium.

There is high uncertainty associated with the selected LOAEL and NOAEL because they are based on only one study that measured the acute effects of short-term exposure to chromium. The chronic effects of chromium on other fish species is unknown and not available from the literature reviewed. Further, there is uncertainty associated with the selected tissue TRVs for chromium because metals are metabolically regulated by fish following exposure. Therefore, whole-body tissue residues will vary based on the duration of exposure, the exposure or uptake route (dietary versus aqueous), whether the fish was pre-exposed to metals, and the rate at which the metal is regulated. Therefore, the measured tissue-residue concentrations do not necessarily reflect critical tissue burden concentrations associated with effects, and the tissue TRVs for chromium may overestimate or underestimate toxicity to fish.

4.2.2.2 Lead

Five toxicity studies that measured effects and whole-body concentrations associated with lead exposure were evaluated for TRV selection (Table 4-15). The lowest chronic LOAEL based on a measured tissue concentration was selected as the LOAEL TRV (lower LOAELs based on injected concentrations were available). At the selected LOAEL of 4.02 mg/kg ww, body weight was significantly reduced in third-generation, 12-week-old brook trout following aqueous exposure to lead (Holcombe et al. 1976). The selected NOAEL TRV of 2.54 mg/kg ww was based on the same study. The selected LOAEL and NOAEL TRVs are similar to, but slightly higher than, the SL TRV of 2.2 mg/kg ww derived from (Dyer et al. 2000)).

There is uncertainty associated with the selected tissue TRVs for lead because metals are metabolically regulated by fish following exposure. Therefore, whole-body tissue residues will vary based on the duration of exposure, the exposure or uptake route (dietary versus aqueous), whether the fish was pre-exposed to metals, and the rate at which the metal is regulated. Therefore, the measured tissue-residue concentrations do not necessarily reflect critical tissue burden concentrations associated with effects, and the tissue TRVs for lead may overestimate or underestimate toxicity to fish.

4.2.2.3 Mercury

Twenty toxicity studies that measured concentrations of mercury and effects associated with mercury exposure were evaluated for TRV selection (Table 4-16). The LOAEL and NOAEL of 0.47 and 0.2 mg/kg ww, respectively, were selected as the tissue TRVs for mercury. These were the lowest measured whole-body tissue LOAEL and NOAEL TRVs based on a direct measure of survival, growth, or reproduction. A lower whole-body tissue LOAEL of 0.23 mg/kg ww was reported in Webber and Haines (2003). At this concentration, predator avoidance behavior was reduced in golden shiners exposed to dietary methylmercury for 90 days. The authors reported that the observed behavioral changes could affect fish survival. However, this has not been confirmed in field studies; and thus, the LOAEL based on this study was not selected. Friedmann et al. (1996) also reported a lower LOAEL; however, this study was not selected because whole-body concentrations lacked viscera and the observed growth effect was not dose-responsive. The selected LOAEL is just below

the 0.5 to 1.0 mg/kg ww range that Beckvar et al. (1996) cited as being concentrations at which reproductive and early life stage effects possibly start to occur in fish. Based on Beckvar et al., the selected LOAEL TRV likely represents an acceptably conservative effect-level threshold for fish. The selected LOAEL TRV is similar to the SL TRV, 0.46 mg/kg ww, which was derived from Dyer et al. (2000).

4.2.2.4 Selenium

The EPA draft chronic ambient water quality criterion for selenium is a whole-body fish tissue concentration. Because different water bodies have different propensities for bioaccumulation of selenium, EPA has issued a draft freshwater aquatic life criterion for long-term effects (the chronic criterion) expressed as a concentration of the pollutant in fish tissue rather than a concentration in the water. The tissue concentration is the most reliable indicator of selenium exposure and risk to fish under different environmental conditions. The draft chronic criterion of 7.91 mg/kg dw was used as the TRV. It was converted from dry weight to 1.58 mg/kg ww assuming 80% moisture content.

4.2.2.5 Zinc

Nine toxicity studies that measured whole-body concentrations of zinc and effects associated with zinc exposure were evaluated for TRV selection (Table 4-17). The LOAEL and NOAEL of 40 and 34 mg/kg ww, respectively, were selected as the whole-body tissue-residue TRVs for zinc (Spehar 1976). The selected LOAEL was the lowest LOAEL TRV reported in the reviewed literature based on adult tissue residues over a chronic exposure period. The selected NOAEL and LOAEL TRV are slightly higher than the SL TRV, 27 mg/kg ww, which was derived from Dyer et al. (2000).

There is uncertainty associated with the selected tissue TRVs for zinc because metals are metabolically regulated by fish following exposure. Therefore, the whole-body tissue residues will vary based on the duration of exposure, the exposure or uptake route (dietary versus aqueous), whether the fish was pre-exposed to metals, and the rate at which the metal is regulated. Therefore the measured tissue-residue concentrations do not necessarily reflect critical tissue burden concentrations associated with effects, and the tissue TRVs for zinc may overestimate or underestimate toxicity to fish.

4.2.2.6 BEHP

Three studies that considered the toxicological effects of BEHP on fish were evaluated for TRV selection (Table 4-18). The LOAEL and NOAEL of 10.6 and 0.39 mg/kg ww, respectively, were selected as the TRVs for all fish receptors based on Mehrle and Mayer (1976). The selected LOAEL is the only LOAEL reported in the reviewed studies. At this selected LOAEL, sac-fry mortality was observed in rainbow trout exposed to BEHP as eggs; whole-body tissue residues were not reported but were estimated using the bioconcentration factors (BCFs; in which the BCF equals the tissue concentration/water concentration) reported for surviving fry

and the associated water concentrations. The calculated sac-fry LOAEL was converted to an adult whole-body LOAEL based on PCB[35] adult-to-sac-fry conversion factors derived from data presented in Niimi (1983) (see Appendix B of the Ecological PRE, Windward 2005). The selected NOAEL TRV is the same as the SL TRV (0.39 mg/kg ww), and the selected LOAEL TRV is two orders of magnitude greater than the SL TRV.

The selected TRVs for BEHP are uncertain because conversion factors were used. There is additional uncertainty associated with the BEHP TRVs because the literature dataset of BEHP toxicity in fish was limited to only one study that reported adverse effects.

4.2.2.7 Total PCBs

Twenty-nine studies on the potential adverse effects of PCB mixtures on fish were reviewed (Table 4-19). The lowest egg concentration was 857 µg/kg ww, which was converted into an adult tissue concentration of 4,020 µg/kg ww using the adult:fertilized egg conversion factor of 4.69 (Table 4-19). This was the lowest acceptable whole-body LOAEL reported in the literature reviewed and was selected as the LOAEL TRV. At the selected LOAEL TRV, Fisher et al. (1994) exposed Atlantic salmon eyed embryos to PCB concentrations of 62.5 to 62,500 µg/L in water for 48 hours, and fry growth was reduced at all exposures. Lower LOAELs were reported; however, these LOAELs were based on field-collected eggs, injection concentrations rather than measured tissue concentrations, or incomplete whole-body tissue concentrations (lacking liver tissue) not associated with the time at which adverse effects were observed. The selected NOAEL (1,900 µg/kg ww) was based on Hansen et al. (1973), in which fry survival was unaffected in sheepshead minnow offspring following maternal exposure for 28 days. This NOAEL was selected as the NOAEL TRV because it was the highest NOAEL reported below the selected LOAEL TRV and was based on a reproductive endpoint. The selected NOAEL and LOAEL TRV are higher than the SL TRV, 720 µg/kg ww, derived from the fifth percentile of all LWG reviewed toxicity studies, and the fifth percentile LOAEL, 800 µg/kg ww, which was derived from Dyer et al. (2000).

There is uncertainty associated with the selected LOAEL TRV because of the application of the egg:adult conversion factor. There is additional uncertainty because Fisher et al. (1994) exposed alevins to high concentrations of PCBs in water (625 µg/L) for a short exposure duration (48 hours), which is not representative of the exposure conditions of the fish receptors that spawn in the Study Area. The uncertainty of the selected LOAEL TRV for PCBs in the context of risk characterization is discussed in the uncertainty analysis (Section 4.6).

[35] A sac-fry to adult tissue concentration conversion factor based on PCB data was used because no data on conversion concentration factors were available for BEHP.

4.2.2.8 Beta-HCH

No relevant literature reporting the toxicity of beta-HCH to fish was identified. Because gamma-HCH (lindane) is structurally similar to beta-HCH, it was selected as a surrogate for beta-HCH. Four toxicity studies that measured the effects of gamma-HCH in water on fish were evaluated for TRV selection (Table 4-20). The selected LOAEL TRV was based on the only study that reported a whole-body tissue residue associated with an adverse effect (Schimmel et al. 1977). An LC50 was reported in sheepshead minnow with tissue-residue concentrations of 79,000 µg/kg ww following acute exposure to lindane in water (Schimmel et al. 1977). This LC50 was selected as the LOAEL TRV.A NOAEL TRV, 1,580 was extrapolated from the LC50 using a UF of 50. The selected NOAEL and LOAEL TRVs are several orders of magnitude higher than the SL TRV, 4.9 µg/kg ww, which was derived from Dyer et al. (2000).

The selected LOAEL and NOAEL TRVs are highly uncertain because they are based on an LC50 following an acute exposure period and were extrapolated from the LC50 using a UF. Other sub-lethal effects likely would occur in fish with lower tissue concentrations. There is additional uncertainty associated with the selected TRVs for lindane because the literature dataset for lindane toxicity to fish is limited to four studies, only one of which measured both whole-body tissue concentrations and observed adverse effects, and lindane (gamma-HCH) was used as a surrogate for beta-HCH. There is additional uncertainty associated with the selected TRVs for lindane because the literature dataset for lindane toxicity to fish is limited to four studies, of which only one measured both whole-body tissue concentrations and observed adverse effects.

4.2.2.9 DDTs

A total of 27 studies that measured DDT concentrations and effects associated with fish exposure to DDT were evaluated for TRV selection (Table 4-21). Only TRVs based on total DDTs or a DDT mixture were considered for TRV selection. TRVs were not developed for individual DDT isomers because insufficient toxicity data were available. The selected LOAEL TRV for total DDTs, 1.8 µg/kg ww, was based on the lowest LOAEL from a controlled study with uncontaminated fish in which a clear dose-response relationship was observed (Allison et al. 1964). In this study, significant mortality was reported for cutthroat trout following 4 months of aqueous exposure to DDT. The selected NOAEL of 1.8 mg/kg ww also was based on Allison et al. This was the highest fish tissue-residue concentration reported at which survival was not significantly affected over 612 days of exposure. The selected NOAEL and LOAEL TRVs (both 1.8 µg/kg ww) are higher than the SL TRV, 0.29 µg/kg ww, which was derived from the fifth percentile LOAEL of all the reviewed fish and invertebrate toxicological studies.

4.2.3 Summary

Tissue EPCs presented in Section 4.2.1 were compared to selected NOAEL and LOAEL TRVs presented in Section 4.2.2 to derive tissue hazard quotients (HQs) for all of the fish tissue Round 2 COPCs: chromium, lead, mercury, selenium, zinc,

BEHP, total PCBs, 4,4′-DDD, 4,4′-DDT, total DDTs, and beta-HCH. The HQ results for the fish tissue-residue assessment are presented in Section 4.6.1.

4.3 DIETARY ASSESSMENT

The dietary dose assessment was one LOE for evaluating risks from metabolized and regulated chemicals to selected fish receptors. In accordance with EPA comments to the Ecological PRE (EPA 2006e), the dietary assessment for fish was based on a dietary dose approach, rather than on estimated dietary concentrations. In the dietary assessment, daily dietary doses were estimated for each fish receptor and dietary chemical pair. Dietary dose estimates included ingestion of biota (prey) and incidental ingestion of sediment. All Round 2 COPC/receptor pairs identified for the dietary dose LOE in Table 4-5 (see Section 4.1.5) were evaluated for the dietary assessment LOE.

4.3.1 Round 2 COPC Dietary Exposure Assessment

This section presents the exposure assessment for the fish Round 2 COPC/receptor pairs identified in Table 4-5. Section 4.3.1.1 presents the approach, Section 4.3.1.2 presents the exposure assumptions, Section 4.3.1.3 presents the sediment and prey tissue concentrations used to calculate exposure doses, and Section 4.3.1.4 presents the estimated dietary doses.

4.3.1.1 Approach

Round 2 COPC exposure estimates were expressed as a dietary dose, representing prey tissue and incidental sediment ingestion. The daily doses were estimated using Equation 4-1.

$$IR_{diet} = \frac{(FIR \times C_{prey}) + (SIR \times C_{sed})}{BW} \times SUF \qquad \textbf{Equation 4-1}$$

Where:

IR_{diet}	=	estimated fish ingestion rate or exposure dose (mg/kg bw/day)
FIR	=	food ingestion rate (kg ww food/day)
C_{prey}	=	tissue concentration in prey items (mg/kg ww)
SIR	=	sediment ingestion rate (kg dw sediment/day)
C_{sed}	=	concentration in surface sediment (mg/kg dw)
BW	=	body weight (kg)
SUF	=	site use factor (unitless); fraction of time that a receptor spends foraging at the site relative to the entire home range

Body weights, food ingestion rates, sediment ingestion rates, and site use factors vary among fish receptors as described in Section 4.3.2.2. Table 4-22 presents the dietary parameters that were used to estimate dietary EPCs and dietary exposure doses for fish, including body weight, food ingestion rate, and sediment ingestion rate. Dietary parameters for fish are the same as those used in the Ecological PRE (Windward

2005), except where noted. Dietary parameters were modified from the Ecological PRE based on EPA comments (EPA 2006b) or additional information regarding incidental fish sediment ingestion.

Body weights were based on the average body weights measured in individual fish from the Round 1 and Round 2 tissue sampling efforts. Measured food ingestion rates were not available for the fish receptors and were estimated using the equation presented in Weininger (1978):

$$FIR = (0.022 \times BW^{0.85}) \times \exp^{(0.06*T)}$$ **Equation 4-3**

Where:

FIR	=	food ingestion rate (mg/kg bw/day)
BW	=	body weight (kg)
exp	=	2.71828
T	=	temperature (degrees Celsius); a temperature of 13.4° C was used in the calculations of fish food ingestion rates based on the average data collected by ODEQ from a station within the Study Area (SP&S railroad bridge) from 1995 to 2005 (ODEQ 2005).

In comments on the Ecological PRE (EPA 2006b), EPA requested that a range of body weights be reported and that implications for the potential range on fish body weights on risk assessment be discussed because body weight is used to estimate food ingestion rates. However, because both the feeding rate and the exposure rate (dietary dose estimate; IR_{dose}) are a function of body weight, the specific body weight used in the calculation is immaterial to the resulting dose and will not have an impact on the risk estimates or conclusions.

Measured sediment ingestion rates were not available for fish receptors and were estimated using the following equation:

$$SIR = (FIR \times F_{solids}) \times F_{sed}$$ **Equation 4-4**

Where:

SIR	=	sediment ingestion rate (mg/kg dw)
FIR	=	food ingestion rate (mg/kg ww)
F_{solids}	=	fraction of food that is dry weight ($F_{solids} = 1 - F_{moisture}$)
F_{sed}	=	fraction of diet that is incidentally ingested sediment

To determine sediment ingestion rates on a dry weight basis, food ingestion rates were first converted to dry weight based on the percent moisture in each receptors prey item. The fraction of sediment in the diet (F_{sed}) was also not available for most fish receptors; therefore, for each receptor where data were not available, F_{sed} was estimated using best professional judgment in consultation with fish biologists who have conducted dietary studies with the receptor species.

The chemical concentration in sediment (C_{sed}) was calculated as the UCL over the sediment exposure area for each receptor. The chemical concentration in each receptor's prey (C_{prey}) was based on concentrations in representative prey items from the ERA dataset (Section 2.0). All Round 2 COPCs were conservatively assumed to have the same bioavailability in the field as in the laboratory toxicity study that provided the basis for the TRV. C_{prey} was calculated as the concentrations in each component of the receptor's diet and the fraction of each component relative to all prey items consumed. For example, the concentrations in food for a receptor that might ingest both fish and benthic invertebrate prey was estimated using the following equation:

$$C_{prey} = (C_f \times F_f) + (C_b \times F_b) \qquad \textbf{Equation 4-5}$$

Where:

C_{prey}	=	concentration in prey items (mg/kg ww)
C_f	=	concentration in fish prey tissue (mg/kg ww)
F_f	=	fraction of the receptor diet consisting of fish (kg fish/kg food)
C_b	=	concentration in benthic invertebrates prey tissue (mg/kg ww)
F_b	=	fraction of the receptor diet consisting of benthic invertebrates (kg benthic invertebrates/kg food)

Table 4-23 presents the prey species in the ERA dataset that were used to represent fish prey items and the portion of the receptor's diet that each prey item comprises to calculate the concentration in prey tissue (C_{prey} in Equation 4-2). The dietary portion of each component in each receptor's diet was based on information from the literature. Prey items included fish and invertebrate species (i.e., largescale sucker, carp, sculpin, peamouth, northern pikeminnow, crayfish, clams) collected in the field in the Study Area, and invertebrate species (i.e., worms) that underwent laboratory bioaccumulation testing. Clams collected in the field were used to represent clam prey for the fish receptors; clams that underwent laboratory bioaccumulation testing were not used in the dietary prey scenarios. The prey items identified for each receptor were agreed upon by EPA, EPA's partners, and LWG in the development of the Ecological PRE (Windward 2005). Prey portion assumptions for fish diets are the same as those used in the Ecological PRE, except where noted. Prey portion assumptions were modified from the Ecological PRE based on EPA comments (EPA 2006b) or on additional data collected since the Ecological PRE (i.e., Round 2 tissue data collection). Not all Round 2 COPCs were analyzed in all prey items. Therefore, surrogate species were selected from similar feeding guilds to represent prey tissue concentrations, as noted in Table 4-23.

4.3.1.2 Dietary Exposure Assumptions

This section describes how the exposure parameters were selected for each fish receptor species, as presented in Tables 4-21 and 4-22. For all fish receptors, a site use factor of 1.0 was assumed, and fish were assumed to forage on sediments throughout the Study Area (C_{sed} was calculated using surface sediment data from

throughout the Study Area). Specific factors affecting sediment exposure areas (e.g., site fidelity, migration, home/foraging range) are discussed in the uncertainty section. To evaluate the uncertainty of assuming a Study Area-wide foraging area for fish receptors with home ranges less than the entire Study Area (i.e., sculpin, smallmouth bass, and northern pikeminnow), a range of sediment concentrations (C_{sed}) and tissue concentrations (C_{tissue}) based on smaller foraging areas and the effects on exposure doses are discussed in the uncertainty analysis (Section 4.6.2).

4.3.1.2.1 Largescale Sucker

Body Weight and Daily Food Ingestion Rate

The average body weight reported for individual largescale suckers collected in Round 1 sampling was 794 g. This body weight was used with Equation 4-3 to estimate a food ingestion rate of 0.0332 kg ww/day for largescale sucker.

Incidental Sediment Ingestion Rate

The largescale sucker prefers to remain close to the bottom in shallow waters, both as a juvenile and as an adult, and is primarily a bottom-feeder. This native fish is known to consume large amounts of sediment during feeding as an adult (CBFWA 1996), and sediment was noted in the stomach content analyses of four largescale suckers collected during Round 1 sampling (Attachment B8 of Appendix B of the Programmatic Work Plan, Integral et al. 2004a). Up to 30% of the gut content was unidentified organic matter for largescale sucker from Lake Roosevelt, Washington (Spotts et al. 2002). Charles Lee, a biologist who analyzed the stomach contents for this report, estimated that based on his observation, 10% or less of the unidentified organic material was likely to be sediment (Lee 2006). Based on the information from Lee (2006) and Spotts et al. (2002), 3% of total diet is a conservative estimate of the dietary sediment fraction. Because sediment consumption has not been measured but large quantities of sediment have been noted in largescale sucker diets, a sediment ingestion rate of 8% was conservatively assumed.[36] Using Equation 4-4 and an assumption of 85% moisture (average percent moisture of invertebrate tissue analyzed in the ERA dataset) in the diet, the estimated sediment ingestion rate for largescale sucker was 0.00398 kg dw/day.

Diet Composition

Largescale suckers live in close association with sediment, and benthic invertebrates are a primary component of their diet. Adult largescale suckers almost exclusively consume benthic organisms, detritus and plant material associated with the bottom, including macrophytes, algae, diatoms, detritus and organic material, crustaceans, aquatic insects, and small mollusks (Carl 1936; CBFWA 1996; Spotts et al. 2002). The stomach contents of four largescale suckers collected during Round 1 sampling were analyzed, and prey items included bivalves (*Corbicula* sp.), chironomids, oligochaetes, bryozoans, gastropods, algae, sediment, and detritus (Attachment B.8 the Programmatic WP) (Integral et al. 2004a). Benthic invertebrate tissue data were

[36] Alternative sediment ingestion rates were explored and are presented in the uncertainty section (Section 4.6.2).

collected in Round 2 sampling; however, plant and algae tissue are not available in the ERA dataset. Therefore, the largescale sucker diet (C_{prey}) was modeled using equal portions of laboratory-exposed worms and field-collected clam tissue (Table 4-23).

4.3.1.2.2 Pre-Breeding White Sturgeon

Body Weight and Daily Food Ingestion Rate

A body weight of 15.9 kg was estimated using an allometric model, based on the range of sturgeon body lengths targeted for Round 3 tissue collection. Targeted pre-breeding sturgeon for Round 3 collection body lengths ranged from 42 to 50 inches (Windward 2006a). A length of 27 inches based on 7-year-old pre-breeding sturgeon reported in Wydoski and Whitney (2003) was used to estimate a body weight of 35 lbs (15.9 kg) using available body length and body mass data from the Fraser River (Coast Angler 2006). The estimated body weight was used with Equation 4-3 to derive an estimated food ingestion rate of 0.663 kg ww/day for pre-breeding white sturgeon.

Incidental Sediment Ingestion Rate

White sturgeon are benthic feeders and are in frequent contact with sediments. Sturgeons identify prey on the bottom surface using their long barbels and then extend their mouths and suck up the prey item (USFWS 1961). In a study of 42 juvenile sturgeon (ranging from 22 to 82 cm fork length) from the lower Columbia River, sand mixed with unidentifiable body parts of invertebrate prey items accounted for about 56% of the total wet mass of stomach contents (Romano et al. 2002). Based on expert judgment, the F_{sed} of 8% was selected.

Based on the above, two sediment ingestion scenarios were developed. In Scenario 1, it was estimated that white sturgeon ingest sediment equal to 8 % (F_{sed}) of the volume of food typically ingested. In Scenario 2, it was estimated that sturgeon ingest sediment equal to 56% (F_{sed}) of the volume of food typically ingested. Using Equation 4-4 and an assumption of 85% moisture (average percent moisture of invertebrate tissue analyzed in the ERA dataset) in the diet, the estimated sediment ingestion rates for pre-breeding sturgeon were 0.00796 and 0.0557 kg dw/day for Scenarios 1 and 2, respectively.

Diet Composition

White sturgeon are opportunistic benthic feeders and may feed on crustaceans (e.g., crayfish), shrimp, mollusks, insects and insect larvae, annelids, and small fish (USFWS 1961; Wydoski and Whitney 2003; Radtke 1966; Beamesderfer and Farr 1997). Juvenile white sturgeon feed on amphipods and mysid shrimp (McCabe et al. 1993; Radtke 1966). Columbia River white sturgeon have been reported to feed almost exclusively on amphipods (primarily on *Corophium* spp.) (Romano et al. 2002). In another study, Columbia River white sturgeon were found to prey on small bottom-dwelling mollusks, snails, insect larvae, and crustaceans (Bajkov 1949). Larger sturgeon have been found to feed on fish (Bajkov 1949), including large fish

such as full-grown sockeye salmon, northern pikeminnow, and sucker (USFWS 1961). Immature white sturgeons in the Lower Columbia River feed primarily on amphipods and occasionally on bivalves (*Corbicula fluminea*), chironomid larvae, and fish eggs (McCabe et al. 1993). In the Bonneville and The Dalles pools of the Columbia River, amphipods (almost exclusively *Corophium* sp.) made up 78 and 72% of subyearling and yearling stomach contents, respectively, based on weight reported in Beamesderfer and Nigro (1992). In the same study, mysids (*Neomysis mercedis*) made up 15% and mollusks (*Corbicula* sp.) made up 12% of the yearling white sturgeon stomach contents by weight. Other items in yearling and subyearling white sturgeons included aquatic insects, oligochaetes, and unidentifiable digested material.

Benthic invertebrate tissue data, including bivalve tissue (*Corbicula* spp.) and worm tissue were collected in Round 2 sampling; however, other benthic prey tissue such as insect larvae and amphipods are not available in the ERA dataset. Therefore, the pre-breeding white sturgeon diet (C_{prey}) was modeled using 88% of laboratory-exposed worm tissue (to represent benthic invertebrates living in sediments) and 12% field-collected clams (Table 4-23).

4.3.1.2.3 Juvenile Chinook Salmon

Body Weight and Daily Food Ingestion Rate

The average body weight reported for individual juvenile chinook salmon collected in Round 1 sampling was 12 g. This body weight was used with Equation 4-3 to estimate the IR_{diet} and a food ingestion rate of 0.000501 kg ww/day for juvenile chinook salmon.

Incidental Sediment Ingestion Rate

Low numbers of benthic invertebrates have been reported in the diet of subyearling chinook salmon, suggesting that they are likely feeding in the water column (Vile et al. 2005). and are thus infrequently exposed to sediments. In addition, stomach content analysis of large numbers of juvenile chinook salmon show no evidence of sediment ingestion (Cordell 2001). However, it was conservatively estimated that juvenile chinook salmon may incidentally ingest sediment equal to 1% (F_{sed}) of the volume of food typically ingested. Using Equation 4-4 and an assumption of 79% moisture (average percent moisture of invertebrate tissue [excluding laboratory-exposed clams and crayfish] and small fish analyzed in the ERA dataset) in the diet, the estimated sediment ingestion rate for juvenile chinook salmon was 0.00000105 kg dw/day.

Diet Composition

Based on the stomach contents of juvenile chinook salmon collected in the LWR (n = 346), water fleas (*Daphnia* sp.), fish, and amphipods (*Corophium* sp.) made up 43%, 32%, and 12% of their diet, respectively, based on wet weight (Vile et al. 2005). LWG Round 2 juvenile chinook stomach content data (n = 20) showed that LWR juvenile chinook prey principally on daphnids, terrestrial insects, and chironomid

larvae (Integral and Windward 2006). In the Columbia River, the diet of juvenile chinook salmon in riverine habitats is predominately aquatic insects, including midges and caddisflies (CBFWA 1996; Rondorf et al. 1990). Daphnid and amphipod tissue are not available from the ERA dataset; however, multiplate samplers were used to collect invertebrates, and amphipods (*Corophium* sp.), chironomids, and daphnids (*Daphnia* sp.) were included in the multiplate tissue analysis and are used to represent juvenile chinook diets in this analysis.

Two prey scenarios were evaluated for juvenile chinook salmon. In Scenario 1, the juvenile chinook salmon diet (C_{prey}) was estimated using only multiplate tissue[37] (Table 4-23; multiplate tissue represented 100% of the diet). In Scenario 2, the juvenile chinook salmon diet concentrations were also represented using the juvenile chinook stomach tissues.

4.3.1.2.4 Sculpin

Body Weight and Daily Food Ingestion Rate

The average body weight reported for individual sculpin collected in Round 1 sampling was 19 g. This body weight was used with Equation 4-3 to estimate the IR_{diet} and a food ingestion rate of 0.000794 kg ww/day for sculpin.

Incidental Sediment Ingestion Rate

Sculpin are small predatory fish that live in close association with sediment and are primarily benthic feeders. As adults, sculpin can burrow up to 36 cm into gravel to forage (Wydoski and Whitney 2003). Londi Tomaro, a biologist who studied prickly sculpin in Alaska, noted that although she has not measured sediment in stomachs, in her experience quantifying prickly sculpin stomach contents, a very small proportion of sculpin had sediment in their stomachs; and those that did had a very small amount (Tomaro 2006b). Prickly sculpin are a common sculpin species in the LWR. Based on this information, it was estimated that sculpin incidentally ingest sediment equal to 5% (F_{sed}) of the volume of food typically ingested. Using Equation 4-4 and an assumption of 79% moisture (average percent moisture of invertebrate tissue [excluding laboratory-exposed clams and crayfish] and small fish analyzed in the ERA dataset) in the diet, the estimated sediment ingestion rate for sculpin was 0.00000834 kg dw/day.

Diet Composition

Adult sculpin consume crustaceans, aquatic insects, snails, fish, fish eggs, and mollusks, while juvenile sculpin feed on aquatic insect larvae (Wydoski and Whitney 2003). Sculpin may also prey on other sculpin (Armstrong et al. 1995; Pfister 2003). The stomach contents of eight sculpin collected during Round 1 sampling were analyzed; and aquatic insects, as well as amphipods, gastropods, and bryozoans, were the predominant prey items present (Attachment B6 of the Programmatic WP)

[37] Note that no multiplate tissue chemistry data were available for PAHs, mercury, and TBT. Chemistry data from worm tissue from the laboratory bioaccumulative tissue for these COPCs were used as a surrogate to represented concentrations in the juvenile chinook salmon diet.

(Integral et al. 2004a). Northcote et al. (1954) studied prickly sculpin in British Columbia lakes. All fish analyzed were less than 87 mm, and fish (redside shiner) constituted greater than 75% of the prickly sculpin diet by volume. Tamaro (2006a) observed prickly sculpin that were greater than 102 mm from Auke Lake in Alaska, and noted that 40% of the fish fed on aquatic insects, 37% fed on fish, 35% fed on mollusks, and 25% fed on plant matter, however, the proportions of food items in gut contents were not quantified. The amount of fish consumed by sculpin increased with the size of the sculpin; and Dolly Varden, stickleback, and small sculpin were the principal fish prey of large sculpins (greater than 129 mm in length).

Sculpin analyzed in Round 1 tissue samples ranged from 92 to 172 mm in length; thus, sculpin at the Study Area are likely to feed on a mixture of benthic invertebrates and small fish. However, fish were not a dominant prey item present in the sculpin stomach contents analyzed from the Study Area. Therefore, the sculpin diet (C_{prey}) was modeled using 40% clams and 40% laboratory-exposed worms (to represent benthic invertebrates living in the sediments), as well as 20% sculpin (Table 4-23).

4.3.1.2.5 Peamouth

Body Weight and Daily Food Ingestion Rate

The average body weight reported for individual peamouth collected in Round 1 sampling was 104 g. This body weight was used to estimate the food ingestion rate, dietary EPCs, and exposure doses. Using Equation 4-3, the estimated food ingestion rate for peamouth was 0.00435 kg ww/day.

Incidental Sediment Ingestion Rate

Peamouth consume some benthic species; and while feeding, they may incidentally ingest sediment. However, benthic species make up only a part of the peamouth diet. Peamouth spend a significant portion of time in the pelagic zone and are not likely to have substantial direct contact with sediment. Therefore, it was estimated that peamouth incidentally ingest sediment equal to 1% (F_{sed}) of the volume of food typically ingested. Using Equation 4-4 and an assumption of 79% moisture (average percent moisture of invertebrate tissue [excluding laboratory-exposed clams and crayfish] and small fish analyzed in the ERA dataset) in the diet, the estimated sediment ingestion rate for peamouth was 0.0000456 kg dw/day.

Diet Composition

Peamouth prey on both benthic and pelagic species. Adult peamouth predominately feed on benthic invertebrates, crustaceans, and small fish (Wydoski and Whitney 2003). Peamouth in the mid-Columbia River were reported to prey most frequently (based on percent occurrence) on periphyton, snails, caddisfly larvae, and midges (Gray and Dauble 2001). Larger peamouth occasionally feed on mollusks but rarely on small fish such as small sculpin (Carl et al. 1959). Terrestrial and aquatic insects were observed in the stomach contents of five peamouth collected and analyzed in Round 1 sampling (Attachment B6 of the Programmatic Work Plan, Integral et al.

2004a). Filamentous algae, sediment, fish remains, bryozoans, and statoblasts were also found in Round 1 peamouth stomach contents.

Based on this analysis, peamouth in the Study Area likely feed on a mixture of benthic invertebrates, pelagic invertebrates, and small fish. The peamouth diet (C_{prey}) was modeled using 40% multiplate tissue[38] (representing pelagic invertebrates), 25% laboratory-exposed worms, and 25% field-collected clams (representing benthic invertebrates), and 10% sculpin (representing small fish) (Table 4-23).

4.3.1.2.6 Smallmouth Bass

Body Weight and Daily Food Ingestion Rate

The average body weight reported for individual smallmouth bass collected from within the Study Area during Round 1 sampling was 372 g. This body weight was used to estimate the food ingestion rate. Using Equation 4-3, the estimated food ingestion rate for smallmouth bass was 0.0155 kg ww/day.

Incidental Sediment Ingestion Rate

Smallmouth bass consume benthic prey and may incidentally consume some sediment. Like peamouth, smallmouth bass spend a significant portion of time in the pelagic zone and are not likely to have substantial direct contact with sediment. Therefore, it was estimated that smallmouth bass may incidentally ingest sediment equal to 1% (F_{sed}) of the volume of food typically ingested. Using Equation 4-4 and an assumption of 74% moisture (average percent moisture of fish tissue analyzed in the ERA dataset) in the diet, the estimated sediment ingestion rate for smallmouth bass was 0.0000404 kg dw/day.

Diet Composition

As a benthopelagic species, smallmouth bass consume fish, crayfish and other crustaceans, mollusks, and worms as adults, and insect larvae and zooplankton as juveniles (George and Hadley 1979; Turner 1966; Wydoski and Whitney 2003). In the main stem of the Willamette River, the following proportions (by volume) of prey items were reported based on the identification of the stomach contents of 114 bass (primarily smallmouth bass): 50% sculpin, 26% crayfish, 18% other fish, 2% insects, < 1% salmonids, and 4% other prey items, including snails, worms, and vegetation (ODFW 2001). One study reported stomach contents of smallmouth bass in fish collected in 2002 and 2003 from the LWR (Pribyl et al. 2005; Vile and Friesen 2005). Crayfish and fish (sculpin) made up 62 and 35% (by wet weight), respectively, of the stomach contents of 48 juvenile smallmouth bass (< 100 mm) (Vile and Friesen 2005). Fish (primarily sculpin) made up approximately 90% of the stomach contents of 17 adult smallmouth bass (> 200 mm) (by wet weight); and shrimp and crayfish made up 7% and 3%, respectively, of the stomach contents (Pribyl et al. 2005). Crayfish remains were the dominant prey item in the stomach contents of four

[38] Note that no multiplate tissue chemistry data were available for PAHs, mercury, and TBT. Chemistry data from worm tissue from the laboratory bioaccumulative tissue for these COPCs were used as a surrogate to represent water column invertebrate concentrations in the peamouth diet.

smallmouth bass collected in Round 1 (Attachment B6 of the Programmatic Work Plan, Integral et al. 2004a). In addition, juvenile chinook salmon were not found in the digestive tracts of smallmouth bass collected from RM 12 of the LWR during peak juvenile salmonid migration from April to June and do not appear to be a typical prey species of smallmouth bass (Fishman 2001).

Representative crayfish and fish prey tissue data for smallmouth bass are available from Round 1 sampling. Based on the data reported in the reviewed literature, smallmouth bass in the Study Area likely feed primarily on sculpin and other small fish. Because the smallmouth bass collected in Round 1 ranged from 220 to 432 mm in length, the prey portions for this receptor were based on the smallmouth bass larger than 200 mm reported in Pribyl et al. (2005), in which small fish (sculpin) made up the majority of the prey items. Therefore, the smallmouth bass diet (C_{prey}) was modeled using 90% sculpin, 5% crayfish, and 5% field-collected clams (Table 4-23).

4.3.1.2.7 Northern Pikeminnow

Body Weight and Daily Food Ingestion Rate

The average body weight reported for individual northern pikeminnow collected in Round 1 sampling was 558 g. This body weight was used to estimate the food ingestion rate. Using Equation 4-3, the estimated food ingestion rate for northern pikeminnow was 0.0233 kg ww/day.

Incidental Sediment Ingestion Rate

As with smallmouth bass, northern pikeminnow may occasionally come into contact with sediment when foraging, though they are not likely to have substantial direct contact with sediment. Therefore, it was estimated that northern pikeminnow may incidentally ingest sediment equal to 1% (F_{sed}) of the volume of food typically ingested. Using Equation 4-4 and an assumption of 74% moisture (average percent moisture of fish tissue analyzed for the ERA dataset) in the diet, the estimated sediment ingestion rate for peamouth was 0.0000606 kg dw/day.

Diet Composition

Northern pikeminnow is a benthopelagic species, consuming predominately fish and some insects (Buchanan et al. 1981; Jeppson and Platts 1959). Fish and crayfish were the dominant prey items in the stomachs of northern pikeminnow collected from the mid-Columbia River, making up 32% and 16%, respectively, of the diet by volume (Gray and Dauble 2001). Crayfish and fish have been found to be dominant prey items of the northern pikeminnow in the Willamette River as well. Fish, crayfish, and insects were present in 36, 31, and 26%, respectively, of northern pikeminnow stomachs that contained prey items for fish collected from locations in the Willamette Basin (Buchanan et al. 1981). In the same study, sculpin, salmonids, and other fish (i.e., cyprinids, catostomids, and lampreys) made up 68, 10, and 22%, respectively, of the total northern pikeminnow fish diet based on stomach contents (Buchanan et al. 1981). In another study, crayfish made up approximately 58% (by wet weight) of the digestive tract contents of northern pikeminnow collected in the LWR (n = 20) from

2002 to 2003 (Pribyl et al. 2005). Other prey made up 42% of the contents in northern pikeminnow digestive tracts. Fish and crayfish remains were found in the stomach contents of four northern pikeminnow collected from the Study Area during Round 1 tissue sampling (Attachment B6 of the Programmatic Work Plan, Integral et al. 2004a).

Based on the site-specific data and the reviewed literature, northern pikeminnow likely feed on a variety of fish species as well as on crayfish and benthic invertebrates in the Study Area. The northern pikeminnow diet (C_{prey}) was modeled using 30% crayfish, 25% sculpin, 25% laboratory-exposed worms, and 5% each of four other fish prey items (largescale sucker, carp, peamouth, and northern pikeminnow) (Table 4-23).

4.3.1.3 Prey Tissue and Surface Sediment EPCs

This section presents the Round 2 COPC concentrations in prey tissue and sediment that were used in Equation 4-2 to calculate dietary doses for each fish receptor. Dietary exposure doses for each Round 2 COPC/receptor pair were based on an estimate of each Round 2 COPC EPC in surface sediment and prey tissue samples. EPCs were calculated as an upper confidence limit (UCL) on the mean following EPA guidance.

UCL sediment and prey tissue EPCs were calculated for all Round 2 COPCs using ProUCL v. 3.0 software, which was developed by EPA to compute an appropriate 95th UCL of an unknown population mean. ProUCL tests for normality, lognormality, and a gamma distribution of the dataset and computes a conservative and stable UCL of the unknown population mean (EPA 2004c). Prior to running data through ProUCL to determine UCLs, non-detected data were adjusted to one-half the detection limit to represent the concentration value[39]. For each Round 2 COPC, the UCL recommended by ProUCL was used as the EPC for the risk calculations. In the case where an insufficient unique number of data values were available, the maximum concentration[40] was used to represent the EPC. Attachment G1 presents the summary statistics (i.e., minimum, maximum, and mean Round 2 COPC concentrations), distribution types, ProUCL-recommended UCLs, and selected EPCs in prey tissue and sediment for each Round 2 COPC.

[39] Use of a regression on order statistic to estimate values to represent non-detected concentrations may be appropriate for calculation of UCLs in the BERA.
[40] Where the maximum concentration was a non-detected value, the full detect limit was used to represent the EPC.

4.3.1.3.1 Prey Tissue

Tissue data from eight prey species collected from the Study Area were used to represent the potential prey of fish receptors: field-collected clams, multiplate invertebrates, crayfish, carp, largescale sucker, northern pikeminnow, peamouth, and sculpin.[41] Juvenile chinook stomach content tissues and laboratory-exposed worm tissue were also used to represent prey tissue concentrations. These data are a part of the ERA dataset, as described in Section 2.0.

For all fish receptors, it was assumed that they consumed prey from throughout the Study Area; therefore, the exposure concentration for each prey item was calculated using all tissue data in the ERA dataset. The EPC for each tissue type was estimated as the UCL. The calculated UCL Round 2 COPC concentrations in each tissue type are presented in Table 4-24.

4.3.1.3.2 Surface Sediment

Surface sediment data from the entire Study Area were used to estimate the Round 2 COPC exposure resulting from incidental sediment ingestion. Round 2 COPC concentrations in sediment were calculated as UCL concentration of all surface sediment samples. The calculated UCL EPCs in sediment are presented in Table 4-25.

4.3.1.4 Estimated Dietary Doses

Exposure dietary dose estimates (based on UCL prey tissue and sediment EPCs) were calculated for each Round 2 COPC/receptor pair based on the information presented in Sections 4.3.2.2 and 4.3.2.3. Estimated dietary dose exposures are presented in Table 4-26.

4.3.2 Round 2 COPC Dietary Effects Assessment

This section presents a brief summary of the toxicity literature reviewed for each of the Round 2 COPCs. NOAEL and LOAEL TRVs were presented in the Ecological PRE (Windward 2005). In the Ecological PRE, both dietary concentration and dietary dose TRVs were calculated for fish. EPA comments on the Ecological PRE requested that only dietary dose TRVs be used for fish and that diet concentration-based TRVs be omitted (EPA 2006c). In response to EPA's comments, only dietary dose TRVs were used; however, it should be noted, that there is uncertainty associated with dietary dose-based fish TRVs because of the limited availability of fish feeding rates and body weights in the toxicological studies. Wildlife dose-based TRVs are frequently used in ERAs, and standard ingestion rates and body weights are available; however, for fish, the dietary dose-based approach is not commonly used in ERAs, and limited data are available to calculate dietary dose TRVs.

[41] Note that the sizes of the fish analyzed for tissue chemistry data are larger than those likely to be eaten as prey. Tissue concentrations of some chemicals (e.g., cadmium (Douben 1989), mercury (Post et al. 1996), and PCBs (Gobas 1993)) increase with fish body size; whereas, tissue concentrations of other chemicals decrease with body size (e.g., zinc (Patrick and Loutit 1978, as cited in Douben 1989)). For some chemicals, body size may not strongly affect tissue concentrations. The uncertainty in fish diet assumptions is evaluated and presented in the uncertainty section (Section 4.6.2).

The assumptions used in the dose-based TRVs presented in the Ecological PRE (Windward 2005) were revisited, and fish ingestion rates were evaluated in greater detail – specifically for those studies in which neither a dietary dose nor a feeding rate was presented in a given toxicological study. For such studies, the dose was estimated using the average feeding rate from toxicological studies for the same species of fish fed a similar food type (i.e., live prey or a synthetic diet). Details on the assumptions and sources of dietary feeding rates and the derivation of updated dietary dose TRVs are presented in the TRV technical memorandum addendum in Attachment G5. Round 2 TRVs were not based on studies that measured effects in field-collected organisms, or that were based on behavior endpoints. Behavior studies were evaluated as part of the TRV review process, per EPA's direction (EPA 2004b). However, for the Round 2 ERA, behavior studies were not considered direct measures of the survival, growth, and mortality endpoints that were selected and agreed upon for this fish ERA (Table 4-3). Toxicity studies based on field-collected organisms are not appropriate for evaluating risks beyond a screening-level evaluation because the presence of multiple chemicals and other environmental factors may result in adverse effects that complicates the interpretation of field study results. In addition, in accordance with EPA comments on the Ecological PRE (EPA 2006c), NOAEL TRVs were extrapolated from unbounded LOAEL TRVs when no toxicological data were available, using UFs based on EPA Region 10 guidance. The use of UFs is highly uncertain and not recommended by LWG for use in determining risks for making risk management decisions.

Dietary dose NOAEL and LOAEL TRVs were selected for all fish Round 2 COPCs. It should be noted that selected dietary dose TRVs for fish may not be appropriate for use in the BERA and may be revisited prior to the preparation of the BERA.

4.3.2.1 Cadmium

Eight studies that reported the effects of dietary cadmium on fish were considered for TRV selection (Table 4-27). The lowest LOAEL of 0.010 mg/kg bw/day, for growth of juvenile rockfish was selected as the LOAEL TRV for cadmium (Kim et al. 2004; Kang et al. 2005). No NOAEL below this selected LOAEL TRV was derived from the reviewed studies, so a NOAEL TRV was extrapolated from the LOAEL TRV using a UF of 5. The resulting NOAEL of 0.0020 mg/kg bw/day, was selected at the NOAEL TRV.

There is some uncertainty associated with the selected dietary fish TRVs for cadmium. The selected LOAEL TRV is two to three orders of magnitude lower than the NOAELs reported in the six other studies (1.0 to 9.4 mg/kg bw/day) and is three orders of magnitude lower than the LOAELs reported in the three other studies that reported LOAELs (4.6 to 68 mg/kg bw/day). As such, the selected TRVs are conservative because the majority of the toxicological studies reviewed indicate that the selected TRVs may over predict cadmium toxicity to fish (by several orders of magnitude). There is additional uncertainty associated with the use of a UF to derive a NOAEL TRV.

4.3.2.2 Copper

Thirteen toxicity studies that reported the effects of dietary copper on fish were evaluated for dietary TRV selection (Table 4-28). The LOAEL and NOAEL of 0.48 and 0.24 mg/kg bw/day, respectively, were selected as the dietary TRVs for copper. These were the lowest TRVs derived from the reviewed toxicological literature. At 0.48 mg/kg bw/day, a reduction in channel catfish fingerlings was observed following exposure to dietary copper for 16 weeks (Murai et al. 1981). At 0.24 mg/kg bw/day, no effect on body weight was observed.

4.3.2.3 Mercury

Five studies that measured the effects of dietary mercury on fish were evaluated for TRV selection (Table 4-29). The LOAEL and NOAEL of 0.048 and 0.013 mg/kg bw/day, respectively, were the selected as the dietary TRVs for mercury. These were the lowest acceptable TRVs reported in a study in which relevant fish measurement endpoints of survival, growth, or reproduction were directly measured. At the selected LOAEL, a decrease in male mummichog survival was observed, and at the selected NOAEL, no effect on male mummichog survival was observed (Matta et al. 2001). A lower LOAEL TRV of 0.0091 mg/kg bw/day was derived from Webber and Haines (2003), in which predator avoidance behavior was reduced in golden shiners exposed to dietary methylmercury for 90 days. The authors contend that the observed behavioral changes could affect fish survival; however, this has not been confirmed in field studies. Therefore, the LOAEL based on Webber and Haines was not selected.

4.3.2.5 Tributyltin

Only one study that measured the toxicity of dietary TBT to fish was reviewed (Shimasaki et al. 2003). Shimasaki et al. reported the effects on reproductive success of Japanese flounder larvae following exposure to dietary TBT (Table 4-30). The flounder larvae that were exposed to TBT were generated from two parents that were genetically female (with XX chromosomes); however, one parent was phenotypically and functionally male, and offspring were generated. All offspring produced were genetically female (XX). The female offspring larvae were exposed to 0.0021 or 0.020 mg/kg bw/day TBT as tributyltin oxide for approximately 65 days, and the effects on growth and sex reversal were measured.

A significant reduction in growth was reported at 100 days at both doses; however, at the lower dose, growth was recovered by 300 days, so this was not considered an adverse effect. An increase in sex reversal was reported in TBT-exposed fish. The actual impact of TBT on sex reversal is unclear because sex reversal was apparent in the parent fish prior to TBT exposure; and sex reversal did not appear to affect the production of offspring. Because the ecological significance of the sex reversal endpoint is uncertain, the LOAEL of 0.020 mg/kg bw/day associated with the growth endpoint was selected as the dietary LOAEL TRV for TBT. The growth NOAEL of 0.0021 mg/kg bw/day was selected as the NOAEL TRV. There is uncertainty associated with the selected TRVs because the literature dataset for the dietary toxicity of TBT to fish is limited to one study.

4.3.2.6 PAHs

Four studies evaluated the toxicity of benzo(a)pyrene to fish (Table 4-31). A LOAEL of 1.4 mg/kg bw/day was selected as the LOAEL TRV for fish because this was the lowest LOAEL reported in the studies reviewed. The NOAEL of 0.66 mg/kg bw/day was also selected. In the selected study, Rice et al. (2000) fed juvenile English sole polychaete worms that were exposed to benzo(a)pyrene spiked sediments. Adverse effects on body weight were observed in the highest dose group (at 1.4 mg/kg bw/day). There is uncertainty associated with the selected TRVs for benzo(a)pyrene because the literature set for the toxicity of benzo(a)pyrene is limited.

Three studies in which fish were exposed to a mixture of PAHs were identified (Table 4-31). A LOAEL of 18.0 mg/kg bw/day was selected as the LOAEL TRV for fish because this was the lowest acceptable LOAEL reported in the three studies reviewed. The NOAEL of 6.1 mg/kg bw/day from this study was also selected. In the selected study, Meador et al. (2006) exposed juvenile chinook salmon to a mixture of 21 PAHs[42] in a prepared diet at a total PAH dose of up to 22 mg/kg bw/day. Adverse effects on weight were observed in the highest dose relative to the control on a wet-weight basis. Adverse effects on weight were also observed at the next highest dose (18.0 mg/kg bw/day) relative to the control when fish weights were considered on a dry-weight basis. A lower LOAEL of 0.12 mg/kg bw/day was derived from Rice et al. (2000) where juvenile English sole were few polychaete worms that had been previously exposed to field-collected sediments from Eagle Harbor, Washington. However, this LOAEL was not selected because the dietary prey (i.e., worms) was exposed to field-collected sediments with uncharacterized chemicals and the significance of the effects observed at this concentration were statistically ambiguous.

There is uncertainty associated with the selected TRVs. The PAH mixture used in the selected study was designed to resemble the PAH mixture observed in juvenile chinook stomach contents collected in the Duwamish River, Seattle (Meador et al. 2006). This mixture may or may not be consistent with PAHs mixtures found in the Study Area. Thus, the PAH mixture TRVs may not represent the actual toxicity of the mixture of PAHs evaluated in the exposure assessment (Section 4.3.2). There is additional uncertainty associated with the selected TRVs because the literature dataset for the toxicity of PAH mixtures to fish is limited to two studies.

4.3.2.7 Total DDTs

Eleven studies that measured the effects of dietary DDT to fish were reviewed (Table 4-32). The LOAEL of 0.14 mg/kg bw/day was selected as the dietary LOAEL TRV for DDT. This selected LOAEL represents the lowest LOAEL reported in the literature reviewed in which fish were exposed to DDT only. At the selected LOAEL,

[42] The PAH mixture included the following chemicals: naphthalene, 2-methylnaphthalene, dimethylnaphthalene, dibenzothiophene, acenaphthene, fluorene, 1,8-dimethyl(9H)fluorene, phenanthrene, 9-ethylphenanthrene, 9-ethyl-10-methylphenanthrene, 1-methyl-7-isopropylphenanthrene, anthracene, fluoranthene, pyrene, methyl pyrene, benz[a]pyrene, benzo(k)fluoranthene, benzo(g,h,i)perylene, dibenzanthracene.

embryo survival was reduced in brook trout following a 156-day exposure of adults (prior to spawning) to a synthetic diet treated with DDT (Macek 1968). A lower LOAEL of 0.0044 mg/kg bw/day was derived from Berlin et al. (1981); however, the field-collected eggs used in this study contained 7.6 mg/kg of PCBs and 4.7 mg/kg of DDE in the tissue and possibly other, uncharacterized organic chemicals that could have contributed to the reported toxicity; and the eggs were exposed to aqueous DDE as well as dietary DDE. Therefore, this LOAEL TRV was not selected.

No NOAEL below this selected LOAEL TRV was derived from the reviewed studies, so a NOAEL TRV was extrapolated from the LOAEL TRV using a UF of 5. The resulting NOAEL of 0.028 mg/kg bw/day was selected at the NOAEL TRV. There is uncertainty associated with the use of a UF to derive a NOAEL TRV.

4.3.2.8 Total PCBs

Nine toxicity studies that measured the effects of dietary PCBs to fish were evaluated for TRV selection (Table 4-33). The LOAEL of 0.050 mg/kg bw/day was selected as the dietary LOAEL TRV for PCBs. The selected LOAEL represents the lowest effect threshold reported in the literature reviewed in which fish were exposed to PCBs only. At the selected LOAEL, reduced fecundity was reported in adult barbels fed 0.050 mg/kg bw/day of Aroclor 1260 in a prepared diet for 50 days (Hugla and Thome 1999). At the selected LOAEL, no adverse reproductive effect was observed in fish fed 0.050 mg/kg bw/day. A lower LOAEL of 0.018 mg/kg bw/day was derived from Berlin et al. (1981); however, the field-collected eggs used in this study contained 7.6 mg/kg of PCBs and 4.7 mg/kg of DDE in the tissue and possibly other, uncharacterized organic chemicals that could have contributed to the reported toxicity; and the eggs were exposed to both aqueous PCBs and dietary PCBs. Therefore, this LOAEL TRV was not selected.

No NOAEL below the selected LOAEL TRV was derived from the reviewed studies, so a NOAEL TRV was extrapolated from the LOAEL TRV using a UF of 5. The resulting NOAEL of 0.0.010 mg/kg bw/day, was selected at the NOAEL TRV. There is uncertainty associated with the use of a UF to derive a NOAEL TRV.

4.3.3 Summary

Dietary dose EPCs presented in Section 4.3.2 were compared to the selected NOAEL and LOAEL TRVs presented in Section 4.3.3 to derive tissue hazard quotients (HQs). The results of this comparison are presented in Section 4.6.1.

4.4 SURFACE WATER ASSESSMENT

In the surface water assessment, the exposure of fish was evaluated by comparing detected chemical concentrations in surface water collected in the Study Area to chemical concentrations in surface water known to be toxic to fish. The surface water Round 2 COPC exposure and effects assessments are presented in Sections 4.4.1 and

4.4.2, respectively. All surface water Round 2 COPCs identified in Table 4-6 (see Section 4.1.5) were evaluated for the surface water assessment LOE.

4.4.1 Round 2 COPC Surface Water Exposure Assessment

All surface water samples were used to estimate fish EPCs (Figure 4-9). It was assumed that all surface water locations represent potential exposure areas for fish receptors foraging throughout the Study Area. Surface water sampling and methods are described in Section 2.1.3.

Surface water EPCs for fish were represented by the UCL of the mean concentration over all surface water samples collected from within the Study Area, from samples collected using both the peristaltic pump and the XAD system. The UCL is a conservative estimate of the average concentration of Round 2 COPCs in surface water and assumed to be representative of surface water exposure concentrations to populations of fish receptors in the Study Area. UCLs were derived using ProUCL software. Prior to running data through ProUCL to determine UCLs, non-detected data were adjusted to one-half the detection limit to represent the concentration value. For each Round 2 COPC, the UCL recommended by ProUCL was used as the EPC for the risk calculations. Attachment G1 presents the summary statistics (i.e., minimum, maximum, and mean Round 2 COPC concentrations), distribution types, ProUCL-recommended UCLs, and selected EPCs in surface water for each Round 2 COPC.

Table 4-34 presents the surface water EPCs for the eleven surface water Round 2 COPCs identified for fish. Total PCB concentrations are represented as the sum of PCB congeners for XAD samples and as the sum of PCB Aroclors for peristaltic samples[43]. Surface water EPCs were compared to the Eco SLs presented in Section 4.4.3.

4.4.2 Round 2 COPC Surface Water Effects Assessment

This section identifies the studies or sources from which the Eco SLs used to evaluate fish Round 2 COPCs were derived. A review of water quality regulatory benchmarks and literature-based thresholds was conducted to develop the Eco SLs for chemicals in water (Attachment G3). Eco SLs were developed by LWG and revised based on EPA's comments (EPA 2006d). The Eco SLs are considered to be protective of all aquatic receptors, including benthic invertebrates, fish, and amphibians. Eco SLs were selected based on EPA's proposed hierarchy proposed hierarchy (LWG 2006a) presented in Attachment G3. In accordance with EPA's comments (EPA 2006d), a UF of 50 was used to calculate a chronic screening value from an acute screening value when no chronic data were available. There is high uncertainty in the use of a UF of 50, and LWG recommends the use of a range of UFs for the BERA. Table 4-35 presents the chronic and acute Eco SLs for surface water Round 2 COPCs.

[43] At six locations where both XAD and peristaltic data were available for PCBs, total PCBs were represented using the total PCB congener sum from the XAD sample.

All acute and chronic Eco SLs for surface water Round 2 COPCs are based on AWQC, with the exception of benzo(a)pyrene, benzo(a)anthracene, and total PCBs. AWQC-based Eco SLs are based on fish and invertebrate toxicity data and designed to be protective of the larger aquatic community. As such these Eco SLs are expected to be protective of the fish populations and communities of the LWR. Brief discussions of the sources of the acute and chronic Eco SLs follow.

Zinc – The Eco SLs for zinc, which are based on AWQC, are hardness-dependent and were adjusted from the AWQC values using EPA-provided equations (EPA 2006f) and a hardness of 25 mg/L calcium carbonate (as estimated for the LWR). Criteria are for the dissolved fraction of zinc, which is the bioavailable fraction.

Benzo(a)anthracene and benzo(a)pyrene – The Eco SLs for these two PAHs were based on Tier II values (Suter and Tsao 1996) derived from 96-hour LC50 daphnid (*Daphnia magna*) studies. No toxicity data specifically for fish were available because PAHs are metabolized by fish, which typically results in very low or non-detectable concentrations of PAHs in tissue. Therefore, the Eco SLs may overestimate or underestimate risks to fish because no fish-specific toxicological data were available.

4-Chloro-3-methylphenol – The acute Eco SLs for 4-chloro-3-methylphenol was obtained from ODEQ's Water Quality Guidance Values Summary (Table 33), which can be used in applying Oregon's Narrative Toxics Criteria (Oregon Administrative Rules [OAR] 340-041-0033(1)) to waters of the state in order to protect aquatic life (ODEQ 2001). The source for this acute criterion is uncertain. In accordance with EPA's comments (EPA 2006d), a UF of 50 was used to extrapolate a chronic screening value from the acute Eco SL because no chronic data were available; however, there is high uncertainty in the use of a UF of 50.

Total PCBs – The acute Eco SL for total PCBs was based on the value (or benchmark) developed by ODEQ and is the same as the Tier II value presented in Suter and Tsao (1996). The acute Eco SL was calculated for AWQC, although it has never been adopted. The chronic Eco SL was based on ODEQ based on partial life-cycle tests conducted with three invertebrate and two fish species. Suter and Tsao cited the same chronic value for total PCBs and noted that it was based on fish residual values.

DDTs – The Eco SLs for 2,4′-DDD, 4,4′-DDD, 2,4′-DDT, 4,4′-DDT, and total DDTs are based on the acute and chronic AWQC for 4,4′-DDT.

4.5 TRANSITION ZONE WATER ASSESSMENT

The assessment of TZW risks for fish receptors is the same as that for benthic invertebrates (Section 3.0), in which exposure was evaluated by comparing detected chemical concentrations in TZW to water concentrations known to be toxic to all

aquatic organisms, including fish (i.e., Eco SLs). All Round 2 COPCs identified for benthic receptors using the TZW framework were evaluated in the TZW assessment for sculpin and lamprey ammocoete receptors (Table 4-7; see Section 4.1.5). These TZW Round 2 COPCs were further evaluated in the benthic risk assessment (Section 3.0) using the TZW framework to arrive at potential iCOCs for benthic invertebrates. The potential iCOCs identified for benthic invertebrates were evaluated in the TZW assessment for sculpin and lamprey ammocoete receptors because of the close associated of these two species (and life stages) with the sediment (Table 4-36).

4.5.1 TZW Exposure Assessment

TZW samples were collected at nine properties throughout the Study Area. Details on the sample locations and a summary of the TZW results are presented in the benthic risk assessment in Section 3.0. The ecological CSM (Figure 4-1) for all fish receptors shows TZW as an incomplete pathway; however, per EPA's request (EPA 2006a), TZW was evaluated as an exposure pathway for sculpin and lamprey ammocoetes. The ecological relevance of TZW to sculpin and lamprey ammocoetes is low. A porewater ventilation rate of 0 to 10% was assumed for sculpin and benthic invertebrates in the FWM (see Appendix E), and the same porewater ventilation rate was assumed for lamprey ammocoetes. The TZW assessment also assumes that organisms are exposed to single TZW sampling locations; however, fish are mobile organisms and the assumption that sculpin and lamprey ammocoetes are exposed to TZW from a single sampling location is highly conservative.

4.5.2 TZW Effects Assessment

A discussion of the Eco SLs is presented in the benthic invertebrate risk assessment (Section 3.0), and Table 3-28 presents the chronic and acute Eco SLs for potential TZW iCOCs. Most of the acute and chronic values for the Round 2 COPCs are based on toxicity studies that examined effects on invertebrate and fish species and therefore are expected to be protective of fish receptors.

4.6 ROUND 2 RISK RESULTS

This section presents the HQs for each LOE. Risk estimates combine the exposure estimates and effects estimates to calculate HQs for each LOE. Critical tissue and dietary dose HQs are presented in Sections 4.6.1.1 and 4.6.1.2, respectively. Surface water and TZW HQs are presented in Section 4.6.1.3 and 4.6.1.4, respectively.

4.6.1 Tissue-Residue Assessment

This section presents risk estimates for fish based on the critical tissue-residue LOE. HQs were derived for each tissue Round 2 COPC/receptor pair using the following equation:

$$HQ = \frac{EPC}{TRV}$$ **Equation 4-6**

Where:

HQ	=	hazard quotient (unitless)
EPC	=	exposure point concentration in tissue (mg/kg ww; represented by a UCL concentration)
TRV	=	NOAEL or LOAEL toxicity reference value in tissue residue (mg/kg ww)

The results of the HQs calculations for the each of the receptors are presented in Tables 4-37 through 4-43 and summarized below. The uncertainty of these risk estimates are presented in Section 4.6.2 to arrive at risk conclusions for fish, presented in Section 4.6.3.

Four Round 2 COPCs (i.e., chromium, total PCBs, total DDTs, and BEHP) were evaluated in largescale sucker whole-body tissue (Table 4-37). NOAEL and LOAEL HQs based on UCL EPCs were all less than 1.0, except for BEHP, where the UCL NOAEL HQ was 1.1. NOAEL and LOAEL HQs were also < 1.0 for all Round 2 COPCs at individual locations. No NOAEL or LOAEL TRV was developed for 4,4′-DDT.

Four Round 2 COPCs (i.e., total PCBs, beta-HCH, total DDTs, and BEHP) were evaluated in measured whole-body sculpin tissue (Tables 4-38 and 4-39) and two Round 2 COPCs (i.e., selenium and BEHP) were evaluated based on predicted tissue concentrations (Table 4-40). The two Round 2 COPCs (i.e., selenium and BEHP) identified through the predictive tissue approach for sculpin were evaluated by comparing the predicted tissue concentration (based on the site-wide UCL sediment concentration multiplied by the BSAF) to the fish NOAEL and LOAEL TRVs to derive HQs. NOAEL and LOAEL HQs based on UCL EPCs were all less than 1.0, except for selenium (based on predicted tissue) and BEHP (based on predicted and empirical tissue). The HQ for the predicted UCL selenium concentration in sculpin tissue was 1.1; however, empirical selenium tissue concentrations did not indicate risk (i.e., selenium was not identified as a Round 2 COPC based on the maximum tissue concentration in sculpin tissue; see Attachment G4). Empirical BEHP concentrations in sculpin tissue were greatest at location 08R003, where the LOAEL HQ was 2.6. The sediment BEHP concentration at location 08R003 was 140 mg/kg TOC and 95th percentile site-wide sediment BEHP concentration was 116.25 mg/kg TOC, so the BEHP HQ based on the empirical tissue dataset is protective of the site. The total DDT LOAEL HQ based on empirical tissue was > 1.0 at one location (07R006) and < 1.0 at all other locations. LOAEL HQs were < 1.0 for total PCBs and beta-HCH at individual locations based on empirical tissue concentrations. No NOAEL or LOAEL TRV was developed for 4,4′-DDT or 4,4′-DDD. Figures 4-10 and 4-11 present the NOAEL and LOAEL HQs for sculpin composites for empirical concentrations of total DDT and BEHP, respectively.

Lead was the only Round 2 COPC evaluated in peamouth tissue (Table 4-41). HQs based on UCL EPC for lead was 9.2. The highest lead concentration was measured at

03R014 and a LOAEL HQ of 2.6 was calculated at this individual sampling location. HQs at all other individual sampling locations were < 1.0.

Zinc was evaluated as Round 2 COPCs in juvenile chinook salmon whole-body tissue (Table 4-42). The NOAEL and LOAEL HQs for zinc based on UCL EPCs were 1.1 and 0.90, respectively. At individual locations, zinc HQs were all less than 1.0 for LOAEL TRVs. No NOAEL or LOAEL TRV was developed for 4,4′-DDT.

Three Round 2 COPCs (i.e., total PCBs, total DDTs, and BEHP) were evaluated in smallmouth bass whole-body tissue (Table 4-43). NOAEL and LOAEL HQs based on UCL EPCs were all less than 1.0, except for BEHP. BEHP concentrations in smallmouth bass tissue were greatest ay location 04R023, where the LOAEL HQs at this individual sampling location ranged from 0.065 to 8.2 (HQs were based on three replicate samples at this location). The total PCB LOAEL HQ was > 1.0 at one location (08R010) and < 1.0 at all other locations. LOAEL HQs were < 1.0 for total DDTs at individual locations. No NOAEL or LOAEL TRV was developed for 4,4′-DDT. Figures 4-12 and 4-13 present the NOAEL and LOAEL HQs for smallmouth bass composites for total PCBs and BEHP, respectively

Three Round 2 COPCs (i.e., total PCBs, mercury, and total DDTs) were evaluated in northern pikeminnow whole-body tissue (Table 4-44). HQs based on UCL EPCs were all less than 1.0, except for mercury, where the UCL NOAEL HQ was 2.0. Mercury concentrations in northern pikeminnow tissue were greatest at location 07R009, where the LOAEL HQ at this individual sampling location was 1.1. LOAEL HQs were < 1.0 for the other Round 2 COPCs (total PCBs and total DDTs) at individual locations for all Round 2 COPCs.

4.6.2 Dietary Dose Assessment

This section presents risk estimates for fish based on the dietary dose LOE. HQs were derived for each Round 2 COPC/receptor pair using Equation 4-6 where the EPC is represented as the estimated ingested dose of Round 2 COPCs (presented in Section 4.3.2) and the TRV is represented by the selected dietary dose NOAEL and LOAEL TRVs (presented in Section 4.3.3). The results of the dietary dose HQs calculations for the each of the receptors are presented in Tables 4-45 through 4-51 and summarized below. The uncertainty of these risk estimates are presented in Section 4.6.2 to arrive at risk conclusions for fish, presented in Section 4.6.3.

Dietary dose LOAEL HQs were < 1.0 for all Round 2 COPCs except TBT and total PCBs. Total PCB LOAEL HQs were > 1.0 for largescale sucker, pre-breeding sturgeon, sculpin, smallmouth bass, and northern pikeminnow. TBT LOAEL HQs were > 1.0 in pre-breeding sturgeon, smallmouth bass, and northern pikeminnow.

4.6.3 Surface Water Assessment

This section presents risk estimates for fish based on the surface water LOE. HQs were derived for each surface water Round 2 COPC using the following equation:

$$HQ = \frac{EPC}{EcoSL} \qquad \textbf{Equation 4-7}$$

Where:

HQ	=	hazard quotient (unitless)
EPC	=	exposure point concentration in surface water (mg/L or µg/L)
Eco SL	=	acute or chronic ecological screening level (mg/L or µg/L)

The results of the HQ calculations for the 10 surface water Round 2 COPCs are summarized below. The uncertainty of these risk estimates are presented in Section 4.6.2 to arrive at risk conclusions for fish, presented in Section 4.6.3.

Acute and chronic surface water HQs are presented in Table 4-52. Acute HQs were < 1.0 for all surface water Round 2 COPCs. Chronic HQs were > 1.0 for the DDTs (as 2,4′-DDT, 2,4′-DDT, 4,4′-DDD, 4,4′-DDT, and total DDTs). Chronic HQs were greatest for total DDTs, where the HQs were 2.4 and 6.7, based on the peristaltic and XAD method, respectively.

4.6.4 Transition Zone Water Assessment

This section presents risk estimates for fish based on the TZW LOE. The TZW HQs calculation results for the potential iCOCs in TZW for fish are the same as those presented for the benthic invertebrate TZW assessment (presented in detail in Section 3.0). The results of the TZW HQ calculations for the potential iCOC are briefly summarized as follows:

- **PAHs** – All individual PAH potential iCOCs exceeded chronic Eco SLs at TZW samples near Gasco and Siltronic properties. At TZW sampling locations near Exxon-Mobil, five PAH potential iCOCs (i.e., benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, fluorene, and phenanthrene) exceeded chronic Eco SLs. Benzo(a)anthracene exceeded chronic Eco SLs at TZW sampling locations near Arco and Kinder Morgan, and benzo(a)pyrene exceeded chronic Eco SLs at TZW samples at Arco.
- **DDTs** – 2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDE, and total DDTs exceeded chronic Eco SLs in TZW samples at the two sites where it was analyzed (i.e., Arkema acid plant and Rhône-Poulenc). 4,4′-DDT exceeded chronic Eco SLs in TZW samples at the Arkema acid plant only.
- **Cyanide** – Cyanide exceeded chronic Eco SLs at the two sites where it was analyzed (Gasco and Siltronic).

- **Perchlorate** – Perchlorate exceeded chronic Eco SLs at two of the three sites where it was analyzed (i.e., Arkema acid plant and Arkema chlordate plant). TZW concentrations at Gunderson did not exceed Eco SLs.

4.7 UNCERTAINTY

This section presents the uncertainties associated with the assumptions and methods for arriving at risk results presented in Section 4.6.1 for each LOE. Uncertainties are presented in the following subsections based on the tissue-residue LOE (Section 4.6.2.1), the dietary LOE (Section 4.6.2.2), the surface water LOE (Section 4.6.2.3), the TZW LOE (Section 4.6.2.4). A summary of the uncertainties are presented in Section 4.6.2.7.

4.7.1 Tissue-Residue Assessment

This section presents the uncertainties associated with the assumptions and methods for deriving HQs for all tissue-residue fish Round 2 COPC/receptor pairs, as presented in Section 4.2.1. Uncertainties are presented in the following subsections based on the exposure assessment, and effects assessment.

4.7.1.1 Exposure Assessment

The following uncertainties associated with the exposure assessment of the tissue-residue LOE were evaluated: 1) the use of this LOE for evaluating risks to regulated metals, 2) the representativeness of the selected ERA fish receptors to other fish receptors in the Study Area, 3) the use of non-detected concentrations to represent maximum tissue concentrations, 4) the low detection frequency of BEHP in fish tissues, 5) the use of fish toxic equivalency factors (TEFs) to evaluate dioxins, furans, and dioxin-like PCBs, 5) the expected different between predicted and empirical tissue data, 6) the lack of TBT chemistry data in fish tissue, and, 7) the use of UCLs to represent exposure concentrations.

4.7.1.1.1 Using the Tissue-Residue Line of Evidence for Metals

The tissue-residue LOE was evaluated for all COIs, except PAHs, including regulated metals. Two metals, lead and zinc, were evaluated as Round 2 COPCs in fish tissue; however, most aquatic organisms have specific uptake, internal transport, sequestration, and depuration mechanisms for regulating metals in their tissues (Meyer et al. 2005). Essential metals are regulated because they are necessary for normal metabolic function, whereas, other metals appear to be regulated because they mimic essential elements and are transported by the same mechanisms (Bury et al. 2003 as cited in Meyer et al. 2005). Therefore, tissue-based TRVs for regulated metals may over or underestimate toxicity to fish.

4.7.1.1.2 Representativeness of ERA Dataset

The ERA dataset includes whole-body tissue for selected ecological fish receptors (including largescale sucker, sculpin, peamouth, juvenile chinook salmon,

smallmouth bass, and northern pikeminnow). Whole-body fish tissue-residue data were also available for three fish species collected for the human health risk assessment: carp, brown bullhead, and black crappie. Maximum whole-body tissue concentrations of fish COIs were compared to aquatic SL TRVs to verify that the ecological fish receptors were representative and protective of the fish species present in the LWR, The results of this comparison are presented in the Ecological PRE (Windward 2005). HQs based on SL TRVs were > than 1.0 for the Round 2 COPCs identified for at least one ecological fish receptor (i.e., largescale sucker, sculpin, peamouth, juvenile chinook salmon, smallmouth bass, and northern pikeminnow), with the exception of cadmium. The SL TRV HQ was 1.2 for cadmium in carp (see Ecological PRE, Windward 2005). Cadmium is likely not a Round 2 COPC for fish populations in the Study Area based on fish tissue residues because the SL TRV HQ was greater than 1.0 only for carp (and < 1.0 for all fish considered ecological receptors), the SL TRV HQ for carp is just over 1.0, and cadmium is a metal that is physiologically regulated by fish.

Based on the derivation of HQs for the human health fish receptors, it appears that the selected ecological receptors are representative and protective of the fish species present in the LWR.

4.7.1.1.3 Maximum Tissue Concentrations of COIs Based on Non-Detected Concentrations

Maximum concentrations, regardless of their detection status were compared to SL TRVs to identify Round 2 COPCs in fish tissue. For some chemicals, the maximum reporting limit (RL) was greater than the maximum detected concentration (Table 4-53); however, these COIs were not retained as Round 2 COPCs, because they were either never detected in a specific fish species, or the maximum detected value was not greater than the SL TRV. Seven COIs (and thirteen COI/receptor pairs) had maximum RLs greater than their respective SL TRV.

Detected concentrations of dibutyl phthalate and hexachlorobutadiene in sculpin were below SL TRVs, so risks from these chemicals are likely negligible to sculpin in the Study Area. Similarly, the maximum detected concentrations of butyl benzyl phthalate and dibutyl phthalate in juvenile chinook salmon were below SL TRVs, so risks from these chemicals are likely negligible.

None of the other COI/receptor pairs with maximum RLs > SL TRVs listed in Table 4-53 were detected in fish tissue, and the SL TRV HQs based on RLs do not greatly exceed SL TRVs (maximum HQ is 2.2), so risks are negligible at concentrations below the existing RLs for these COI/receptor pairs.

4.7.1.1.4 Low Detection Frequency for BEHP in Receptor Tissue Samples

BEHP was identified as a Round 2 COPC for largescale sucker, sculpin and smallmouth bass; however, detection frequency of BEHP was low for each of these receptors. BEHP was detected in 33% (2 of 6), 12% (3 of 26), and 14% (2 of 14) of samples for largescale sucker, sculpin, and smallmouth bass, respectively. EPCs

based on UCLs were calculated assuming one-half the RL for non-detected samples, and risks to BEHP may be over or underestimated depending on the actual BEHP concentrations relative to one-half the RL.

4.7.1.1.5 Evaluation of Dioxins, Furans, and Dioxin-Like PCB Congeners Using Fish Toxic Equivalency Factors

The screening step for Round 2 COPC identification for tissue residues compared tissue concentrations of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) to a SL TRV (Section 4.2.1). 2,3,7,8-TCDD was the only dioxin or furan evaluated and it was not identified as a Round 2 COPC in fish tissue based on this comparison.

As an uncertainty analysis, a PCB TEQ and dioxin TEQ (including both dioxins and furans) was calculated for all ecological fish receptors where dioxins, furans, and PCB congeners were analyzed in the tissue (i.e., carp, sculpin, juvenile chinook salmon, and smallmouth bass). TEQs were calculated from the 1998 World Health Organization (WHO) fish TEFs (Van den Berg et al. 1998). Uncertainties in the TEQ approach for evaluating risks associated with dioxin-like PCB congeners and dioxins and furans include limitations in the underlying data used to derive TEFs, the relevancy of the endpoints in the studies, and possible interspecies variability Generally the TEFs for fish are based on in vivo or in vitro studies and the ecological relevance of the TEF approach is lower than the mammalian TEFs.

Attachment 1 presents the fish TEFs. The TEFs quantify the toxicity of each individual dioxin, furan and dioxin-like PCB congener relative to 2,3,7,8-TCDD. These factors and the concentrations of individual dioxins, furans and dioxin-like PCB congeners were then summed to calculate a TEQ, as follows:

$$TEQ = \sum_{i=1}^{n} C_i TEF_i \qquad \textbf{Equation 4-8}$$

Where:

TEF_i = TEF for an individual dioxin, furan, or PCB congener
C_i = tissue concentration of individual dioxin, furan, or PCB congener (pg/g ww)

The PCB TEQ was estimated by summing the TEF concentration products for individual dioxin-like PCB congeners. The dioxin TEQ was estimated by summing the TEF-concentration products for individual dioxins and furans. The results of the comparison of the maximum PCB TEQ and dioxin TEQ concentrations to the SL TRV for 2,3,7,8-TCDD are presented in Table 4-54. Based on this analysis dioxins, furans, and dioxin-like PCB congeners screened out as Round 2 COPCs in fish tissue (i.e., all HQs are < 1.0).

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4.7.1.1.6 Lack of Fish Tissue TBT Data

TBT chemistry data in whole-body fish tissue were unavailable in all fish receptors except for juvenile chinook salmon. TBT does not biomagnify in aquatic systems because dietary uptake in higher trophic level organisms appears to be counteracted by biotransformation in the liver ((Fent 1996) as cited in (EPA 2000a)). TBT tends to accumulate in sediments (Becker van Slooten and Tarradellas 1995), thus, porewater TBT concentrations likely exceed overlying water concentrations and infaunal invertebrates such as clams and worms would almost certainly have higher tissue burdens than fish. Therefore, in this analysis, the available invertebrate TBT tissue chemistry data was compared to fish tissue-residue NOAEL and LOAEL TRVs as a conservative estimate of risk to fish for the tissue-residue LOE.

Three toxicological studies that evaluated the toxicity of TBT to fish were reviewed (Shimasaki et al. 2003; Nirmala et al. 1999; Nakayama et al. 2005) to derive tissue TRVs. Critical tissue residues of TBT were reported in Japanese flounder larvae and Japanese medaka associated with adverse effects on growth and reproductive success, respectively, following exposure to TBT in water. Whole-body tissue-residue[44] LOAELs ranged from 159 µg/kg ww for reduced body weight in Japanese flounder larvae following 65 days of dietary exposure (Shimasaki et al. 2003) to 2,390 µg/kg ww for reduced hatchability and early-life stage mortality of Japanese medaka offspring spawned from exposure of both parents to dietary TBT for three weeks during reproduction (Nirmala et al. 1999). The lowest NOAEL and LOAEL reported in the studies reviewed, 18 and 159 µg/kg ww, respectively, were selected as the most conservative TRVs to evaluate TBT toxicity in fish using tissue whole-body tissue concentrations. There is significant uncertainty associated with this LOAEL TRV because there was no replication of test groups and high mortality was observed consistently across the control and low- and high-dose groups.

Detected TBT concentrations in field-collected clam tissue ranged from 2.5 µg/kg ww to 530 µg/kg ww with a UCL of 220 µg/kg ww. Detected TBT concentrations in laboratory worm data ranged from 2.1 µg/kg ww to 1,700 µg/kg ww with a UCL of 540 µg/kg ww. LOAEL HQs calculated based on the field clam and laboratory worm UCL concentrations were 1.4 and 3.3, respectively. NOAEL HQs calculated based on the field clam and laboratory worm UCL concentrations were 12 and 3.4, respectively. Based on this evaluation, there is likely low risk to fish from TBT; however, there is considerable uncertainty in both the exposure and effects data. Collection of additional fish tissue data would likely result in lower risk estimates; however, because the effects data are uncertain, it is not clear if additional tissue data would reduce uncertainty in the risk characterization.

4.7.1.1.7 Use of UCLs to Represent Tissue EPCs

UCLs were used to represent measured and predicted EPCs for fish. The UCLs were used because they were more conservative than the 80th percentile (i.e., HQs based on

[44] Whole-body tissue concentrations were reported in the reviewed study or were modeled from an egg concentration using an egg:adult conversion factor reported in Nirmala et al. (1999).

UCLs were greater than HQs based on 80th percentiles). However, the 80th percentile concentration is generally recommended to evaluate a population-level endpoint when exposure varies across individuals in the population. Therefore, the use of the UCL of tissue concentrations to represent exposure concentrations for receptors may overestimate risks to fish populations.

4.7.1.2 Effects Assessment

The primary uncertainties associated with the effects assessment of the tissue LOE are the selected NOAEL and LOAEL TRVs. In addition, COIs with no toxicological literature could not be evaluated.

4.7.1.2.1 Selected TRVs

Fish tissue Round 2 COPCs were evaluated using LWG recommended NOAEL and LOAEL TRVs and there are uncertainties associated with the selected TRVs. In general, the uncertainties include the following:

- Few of the laboratory toxicological studies used to derive TRVs were conducted using fish species present in the Study Area. The similarity between the surrogate laboratory and site-specific species is not known.
- The laboratory studies on which TRVs are based were conducted in controlled settings using single-contaminant exposures. Effects associated with multiple-chemical exposure and other environmental stressors present in the Study Area (e.g., habitat loss) were not factored into these studies. It is unknown if these factors would result in additive, synergistic, antagonistic, or neutral effects on overall risk conclusions.
- LOAELs based on egg tissues were converted into adult whole-body tissue concentrations using egg-to-adult conversion factors.
- Limited toxicological data were available for some Round 2 COPCs and selected TRVs were based on a limited dataset.
- NOAELs were extrapolated from LOAELs using UFs, when no NOAEL was available. The use of UFs adds a high degree of conservatism and may overestimate risks. The use of UFs is highly uncertain and not recommended by LWG for use in determining risks for making risk management decisions.
- Selected TRVs based on a surrogate chemical may over- or under-estimate the toxicity of the Round 2 COPC.

The uncertainties of the selected TRVs for each Round 2 COPC are summarized in Tables 4-58 through 4-63, presented at the end of this section.

4.7.1.2.2 COIs with no TRVs

Of the COIs identified in fish tissue, two were not evaluated in the Round 2 COPC screen because no data were available for the development of TRVs. No toxicological studies were identified for benzyl alcohol and butyltin. The risk to fish receptors associated with benzyl alcohol for the critical tissue LOE is unknown because no toxicological data are available. TBT is the most toxic butyltin, and the assessment of TBT was assumed to be protective of fish with regard to the other butyltins (i.e., butyltin ion). Chemical-specific TRVs in the assessment of Round 2 COPCs were also not identified for 2,4′-DDD, 2,4′-DDE, 2,4′-DDT, however, these chemicals were addressed through the risk analysis for total DDTs.

4.7.2 Dietary Dose Assessment

This section presents the uncertainties associated with the assumptions and methods for deriving HQs for all dietary dose Round 2 COPCs, as presented in Section 4.2.2. Uncertainties are presented in the following subsections based on the exposure assessment, and effects assessment.

4.7.2.1 Exposure Assessment

The uncertainties associated with the exposure assessment of the dietary dose LOE for fish receptors include the following: 1) a non-species specific allometric model was used to develop food ingestion rates for all fish receptors, and 2) assumptions that all fish receptors would forage in sediments and tissues throughout the 9-mile Study Area. Both of these uncertainties are discussed and evaluated in the following subsections.

4.7.2.1.1 Use of a Generic Model to Develop Food Ingestion Rates

Measured food ingestion rates were not available for the fish receptors and were estimated using a non-species specific allometric equation developed by Weininger (1978) as presented in Section 4.2.2. Because fish vary widely in their feeding preferences, activity levels, and internal digestion processes, their food requirements likely vary on a species-specific basis. Therefore, food ingestion rates estimated from a non-species specific equation may over or underestimate fish receptor food ingestion rates and thus exposure rates and risk predictions.

4.7.2.1.2 Site-Wide Exposure Scale

For all fish receptors in the dietary dose LOE, it was assumed that each receptor would forage throughout the Study Area (RM 2.0 to RM 11) and consume surface sediment and prey tissue from within this 9-mile reach. This site-wide scale is a reasonable assumption for fish whose home range is equal to or greater than the Study Area (i.e., largescale sucker, pre-breeding sturgeon, peamouth, and juvenile chinook salmon). However, for those fish receptors with home ranges smaller than the Study Area (i.e., sculpin, smallmouth bass, and northern pikeminnow), a smaller foraging range assumption may be appropriate, as supported by the literature.

Sculpin home ranges have been reported in the literature to be as small as 100 ft (30 m) (Hill and Grossman 1987; Natsumeda 1998, 1999). Pribyl et al. (2005) conducted a study from 2000 to 2003, in which the movement of predatory resident fish (including smallmouth bass and northern pikeminnow) was tracked using radio-tagged fish. Similarly, region-specific studies have supported small home ranges for smallmouth bass and northern pikeminnow. Radio-tagged smallmouth bass tended to stay near release points, and the median of the maximum distance traveled by smallmouth bass was 2.3 km (1.4 miles) from the release site over the tracking period; however, most smallmouth bass traveled only 0.4 km (0.25 miles) within 1 month after the release. Radio-tagged northern pikeminnow tended to stay near release points, and the median of the maximum distance traveled by northern pikeminnow was 1.4 km (0.87 miles) from the release site over the tracking period. In addition, all radio-tagged smallmouth bass and the majority of northern pikeminnow collected from the lower portion of the Willamette River (from RM 0.0 to RM 22.5) were located within 20% of the width of the river from either shore, suggesting a preference for nearshore habitat.

Based on the information gathered in the literature, an uncertainty analysis was conducted to evaluate whether reducing the spatial scale of exposure for sediment and prey consumption for sculpin, smallmouth bass, and northern pikeminnow would affect HQs. To evaluate whether reducing the scale of exposure to less than the Study Area for *sediment ingestion* of sculpin, smallmouth bass, and northern pikeminnow would significantly change HQs, the maximum detected sediment concentration of Round 2 COPCs were used to calculate dietary dose HQs for these three fish receptors, to determine an upper range estimate on what HQs could be based on sediments from smaller exposure areas. For most Round 2 COPC/receptor combinations, the use of the single highest detected sediment concentration of the Study Area instead of the site-wide sediment UCL results does not significantly change HQs for all fish receptors. However, there were several chemical/receptor combinations for which changes did occur:

- For sculpin, using the maximum detected sediment concentration resulted in new LOAEL HQ exceedances for cadmium, copper, and TBT. In addition, a new NOAEL HQ exceedance occurred for total DDTs.
- For smallmouth bass and northern pikeminnow, a new NOAEL HQ exceedance for cadmium occurred when the maximum detected sediment concentration was used.

Based on this assessment, it was determined that reducing sediment exposure scales to less than site-wide generally would not affect the HQ results for fish receptors. However, several changes in HQ results could result in additional Round 2 COPCs exceeding LOAEL TRVs for sculpin (copper, cadmium, and TBT).

A similar uncertainty analysis was conducted to evaluate whether reducing the scale of exposure for prey consumption for sculpin, smallmouth bass, and northern pikeminnow would affect HQs. The maximum prey tissue concentrations were used to calculate dietary dose HQs for these three fish receptors, to determine an upper range estimate of what HQs could be based on prey tissues from smaller exposure areas. Using the highest tissue concentrations of prey items within the Study Area instead of tissue EPCs based on site-wide tissue UCLs results in three new HQ exceedances (HQ > 1.0), and changes the magnitude of some HQ. The Round 2 COPC/receptor pairs where new HQ exceedances were calculated are summarized below:

- For sculpin, a new LOAEL HQ exceedance occurred for copper and TBT, and a new NOAEL HQ exceedance occurred for total DDTs.
- For northern pikeminnow, a new NOAEL HQ exceedance occurred for cadmium and total DDTs.

Based on this quantitative analysis, it was concluded that reducing prey tissue exposure scales (i.e., sediment Round 2 COPC concentrations used to estimate dietary exposure dose) to less than site-wide for smallmouth bass and northern pikeminnow generally would not affect the HQ results for fish receptors. However, several changes in HQ results could result in additional Round 2 COPCs (i.e., copper and TBT) exceeding LOAEL TRVs for sculpin.

4.7.2.1.3 Dietary Prey Assumptions

The following uncertainties associated with the dietary prey assumptions were evaluated in the following subsections: 1) the availability of representative prey tissue, 2) the limited number of prey tissue analyzed for TBT, 3) the assumed portions of dietary prey, 4) the assumed sediment ingestion rate assumptions, and, 5) the use of UCL to represent tissue and sediment EPCs.

4.7.2.1.4 Representative Prey Tissue

Representative prey tissue was available for most fish species from the ERA dataset. However, where prey tissue data were unavailable, surrogate prey tissue was used, which may over- or underestimate exposure for some fish receptors based on dietary dose estimates. However, because seven different fish receptors were evaluated, incorporating a range of prey preferences, taken as a whole the dietary exposure scenarios capture the likely range of exposures to fish in the Study Area. The following uncertainties associated with the representative prey items for each of the fish receptors is presented below:

- The largescale sucker diet was estimated using 50% worms and 50% clams. Algae and detritus are both likely major components of the pre-breeding sturgeon diet; however, tissue data for these food items are not available from the Study Area.

Therefore, the modeled diet (clams and worms) for largescale sucker may over- or underestimate chemical dietary exposure doses.

- The pre-breeding sturgeon diet was estimated using 88% worms and 12% clams. Infaunal and epibenthic insect larvae and amphipods are likely another primary component of the pre-breeding sturgeon diet; however, tissue data are not available for these species. Therefore, the modeled diet (primarily laboratory-exposed worms) for pre-breeding sturgeon may over- or underestimate chemical dietary exposure doses.
- The sculpin diet was estimated using 40% worms, 40% clams, and 20% sculpin. Sculpin may prey on other benthic invertebrate prey items or other small fish. However, the modeled diet for sculpin based on sculpin prey and worm and clam tissue prey to represent small fish and benthic invertebrate prey, respectively, is expected to be conservatively representative of the actual dietary exposure dose of sculpin. By including both worms and clams at such high percentages in the diet, it is likely the diet overestimates chemical dietary exposure doses.
- The peamouth diet was estimated using 40% epibenthic tissue from the multiplate samplers (or laboratory-exposed worms as surrogate for PAHs, TBT, and mercury), 25% clams, 25% worms, and 10% sculpin. Peamouth may prey on other benthic or water-column invertebrate prey or other small fish as a benthopelagic feeder. However, the modeled diet for peamouth based on sculpin prey and worm and clam tissue prey to represent small fish and invertebrate prey, respectively, is expected to be conservatively representative of the actual dietary exposure dose of peamouth.
- The juvenile chinook salmon diet was estimated using epibenthic invertebrate tissue from the multiplate samplers data (or laboratory-exposed worms as a surrogate for total PAHs, mercury, and TBT) and juvenile chinook stomach contents tissue for total PAHs, total PCBs, and total DDTs. Juvenile chinook salmon likely prey on a greater portion of water-column organisms that were collected on multiplate samplers (invertebrates collected on multiplate samplers were primarily benthic invertebrates by biomass). Therefore, the modeled diet for juvenile chinook salmon may overestimate or underestimate actual dietary exposure doses.

- The smallmouth bass diet was estimated using 90% sculpin, 5% crayfish, and 5% worms. The smallmouth bass diet is based on a LWR-specific study (Pribyl et al. 2005) and consistent with the diet of other smallmouth bass of the same size class as the fish collected from the Study Area. The modeled diet for smallmouth bass is thought to be likely representative of the actual diet of smallmouth bass in the Study Area.
- The northern pikeminnow diet was estimated using 30% crayfish, 25% sculpin, 25% worms, and 5% of each of the following fish receptors: largescale sucker, carp, peamouth, and northern pikeminnow. The portions of fish species consumed by northern pikeminnow are largely unknown based on the reviewed literature. In addition, northern pikeminnow may consume a greater portion of crayfish. The modeled diet for northern pikeminnow may overestimate or underestimate actual dietary exposure doses; however, selected prey items (i.e., sculpin and worms) were chosen to represent significant portions of the northern pikeminnow diet because dietary fish Round 2 COPCs are generally higher in these two prey items and therefore, may conservatively represent tissue concentrations of other prey items.

4.7.2.1.5 Limited TBT Prey Tissue Data

In the dietary dose assessment, TBT data were available from a limited number of prey species. Butyltins were analyzed for in field clams, and worms that underwent bioaccumulation testing; however, Round 1 fish and crayfish tissue and Round 2 epibenthic invertebrate tissue from multiplate samplers were not analyzed for butyltins. The TBT tissue concentrations of laboratory-exposed worms were used to represent TBT concentrations in prey where TBT was not analyzed, including fish, crayfish, and water-column invertebrates (represented by multiplate epibenthic invertebrate tissue). Therefore, the estimated dietary dose for TBT in those fish receptors consuming fish, crayfish, and/or water-column invertebrate tissue (represented by multiplate tissues), are highly uncertain, and the TBT estimates of risk may be over or underestimated for these fish receptors (i.e., sculpin, peamouth, juvenile chinook salmon, smallmouth bass, and northern pikeminnow).

4.7.2.1.6 Assigned Dietary Prey Fractions

In order to develop dietary dose risk calculations, dietary fractions of prey items used were assigned for each fish receptor (Table 4-23) based on literature searches and best professional judgment, as presented in Section 4.3.2. However, there is some uncertainty associated with the dietary fractions of prey items assigned to each fish receptor (i.e., whether dietary fractions of each prey item should be greater or less than the fraction assigned for estimating risk calculations). Therefore, the impact of changing dietary portion for each prey item on risk estimates (NOAEL and LOAEL HQs presented in Tables 4-44 though 4-51 was evaluated for each fish receptor).

This uncertainty analysis proceeded in a stepwise manner for all dietary Round 2 COPCs for fish, except for TBT due to the lack to data in dietary prey (as discussed in the previous subsection). An initial calculation was conducted for each fish receptor where it was assumed each receptor consumed the relevant prey item with the highest tissue EPC[45] for a given chemical. For those chemical/receptor combinations presented in Table 4-55, using 100% ingestion of the prey items with the highest tissue EPC either: 1) resulted in a new NOAEL or LOAEL HQ > 1.0, or 2) caused a NOAEL or LOAEL HQ exceedance to double in magnitude. For these receptors-chemical pairs, further evaluation was conducted and is described below. For all other chemical/receptor pairs, varying the prey portions resulted in similar NOAEL and LOAEL HQs, and therefore the dietary prey assumptions were assumed to be adequately conservative of Round 2 COPC/receptor pairs in the Study Area.

For those chemical/receptor pairs presented in Table 4-55, dietary prey portions were manipulated to determine the prey species with the highest prey concentration that resulted in an HQ > 1.0 or resulted in an HQ exceedance doubling in magnitude. Dietary fractions were altered to increase the consumption of the prey item with the maximum tissue EPC of a given chemical and lower the prey items with the lowest concentrations. Table 4-55 presents the results of this uncertainty analysis. For each chemical/receptor combination, a "likelihood of occurrence" was assigned as high, moderate, or low. If the dietary fraction required only a small change for a new exceedance and/or there was high uncertainty on prey portions based on the literature (i.e., no region specific studies were available), it was considered to have a high or moderate likelihood of occurrence. However, if the dietary fraction requires a significant change to change HQs and the change was not realistic based on the dietary data provided in the reviewed literature, the assumed dietary prey portions were considered adequately conservative enough for the protection of fish receptors in the Study Area.

Overall, changes to dietary prey portions that were significant enough to cause change to HQs were considered unlikely to occur ("low" or "moderate" likelihood of occurrence) in the Study Area with one exception; for pre-breeding sturgeon, a small increase (7%) in the consumption of clam (from 12% to 19%) would result in a NOAEL HQ > 1.0 for copper.

4.7.2.1.7 Sediment Ingestion

Incidental sediment ingestion rates were not available for fish receptors and were based on best professional judgment, in consultation with fish biologists who have conducted dietary studies with the selected fish receptor species. There is some uncertainty in these assumptions, and assumed sediment ingestion rates may overestimate or underestimate dietary risks to fish, depending on the actual sediment ingestion rates of fish in the Study Area.

[45] EPC based on ProUCL generated value (see Table 5-25).

An evaluation was conducted to determine how modifying the assumed sediment ingestion rates would affect HQs (i.e., whether NOAEL and/or LOAEL HQs would exceed 1.0). First, it was assumed that all Round 2 COPC/receptor pairs would consume the same amount of sediment as food (SI = 100%). Then, the assumed amount of sediment ingested was lowered for each individual pair to determine what percentage of sediment ingestion would result in an HQ > 1.0. Table 4-56 presents the results of this evaluation, listing the fish Round 2 COPC/receptor pairs that would be affected by assuming a higher percentage of sediment ingested as well as the percentage of sediment that would have to be ingested to change the risk conclusions.

The likelihood of occurrence of fish ingesting the amount of sediment required to change risk results for the Round 2 COPCs listed in Table 4-56 is moderate or low.

4.7.2.1.8 Use of UCL to Represent Tissue and Sediment EPCs

In dietary dose calculations, UCLs of prey tissue and sediment concentrations were used to represent EPCs. The UCL is a conservative estimate of the true population mean, and differences between the sample population mean and UCL decrease with data homogeneity and sample size. To evaluate whether the sample means for prey tissue and sediment concentrations result in substantially different risk calculations, dietary dose estimates were calculated using mean concentrations and HQs based on UCL EPCs were compared to HQs based on mean EPCs.[46] Table 4-57 shows this comparison of HQs based on mean EPCs to HQs based on UCL EPCs for all Round 2 COPC-fish receptor pairs where an HQ > 1.0.

For metals, there was generally little change in the HQ when the mean value was used to represent the EPC. However, dietary doses calculated using mean EPCs resulted in the following changes to HQs based on dietary doses calculated using UCL EPCs:

- Copper NOAEL HQs were < 1.0 for peamouth, juvenile chinook salmon, and northern pikeminnow
- Total DDT NOAEL HQs < 1.0 for smallmouth bass
- Total PCB LOAEL HQs < 1.0 for sculpin, smallmouth bass, and northern pikeminnow, and pre-breeding sturgeon, and NOAEL HQs < 1.0 for juvenile chinook salmon

The use of the mean value to represent prey tissue and sediment EPCs also resulted in lower TBT HQs, but generally did not change risk conclusions (whether LOAEL and/or NOAEL HQs were greater than 1.0).

[46] Only HQs based on UCL EPCs greater than one were compared to HQs based on mean EPCs. HQs < 1.0 based on UCL EPCs would also have HQs < 1.0 based on mean EPCs.

4.7.2.2 Effects Assessment

The primary uncertainties associated with the effects assessment of the tissue LOE are the selected NOAEL and LOAEL TRVs. In addition, COIs with no toxicological literature could not be evaluated.

4.7.2.2.1 Selected TRVs

Fish tissue Round 2 COPCs were evaluated using LWG recommended NOAEL and LOAEL TRVs and there are uncertainties associated with the selected TRVs. In general, the uncertainties include the following:

- Few of the laboratory toxicological studies used to derive TRVs were conducted using fish species present in the Study Area. The similarity between the surrogate laboratory and site-specific species is not known.
- The laboratory studies on which TRVs are based were conducted in controlled settings using single-contaminant exposures. Effects associated with multiple-chemical exposure and other environmental stressors present in the Study Area (e.g., habitat loss) were not factored into these studies. It is unknown if these factors would result in additive, synergistic, antagonistic, or neutral effects on overall risk conclusions.
- Actual dietary doses of Round 2 COPCs may vary from those reported in toxicological studies when researchers do not account for the amount of food ingested and the nutritional quality of the food.
- Limited toxicological data were available for some Round 2 COPCs and selected TRVs were based on a limited dataset.
- NOAELs were extrapolated from LOAELs using UFs, when no NOAEL was available. The use of UFs adds a high degree of conservatism and may overestimate risks. The use of UFs is highly uncertain and not recommended by LWG for use in determining risks for making risk management decisions.

The uncertainties of the selected TRVs for each Round 2 COPC are summarized in Tables 4-58 through 4-63, presented at the end of this section.

4.7.2.2.2 COIs with no TRVs

Of the COIs identified for the fish dietary dose LOE, four were not evaluated in the Round 2 COPC screen because no data were available for the development of TRVs. No dietary toxicological studies were identified for antimony, chromium, nickel, and thallium. The dietary risk associated with these chemicals to fish receptors for is unknown because; however, all of these COIs were evaluated using the tissue-residue LOE and were not identified as Round 2 COPC, and therefore, no risk to fish from these chemicals is expected.

4.7.3 Surface Water Assessment

This section presents the uncertainties associated with the assumptions and methods used to derive surface water HQs for fish Round 2 COPCs. Uncertainties are presented in the following subsections based on the exposure assessment, and effects assessment.

4.7.3.1 Exposure Assessment

The following uncertainties associated with the exposure assessment in the surface water LOE were evaluated: 1) the use of UCLs to represent exposure concentrations 2) the use of surface water data based on two sampling methods (i.e., peristaltic and XAD), and 3) the use of NJ-qualified data.

4.7.3.1.1 Use of UCLs to Represent Surface Water EPCs

UCLs were used to represent surface water EPCs for fish across all surface water samples. The UCLs were used because they were more conservative than the 80th percentile (i.e., HQs based on UCLs were greater than HQs based on 80th percentiles). However, the 80th percentile concentration is generally recommended to evaluate a population-level endpoint when exposure varies across individuals in the population. Therefore, the use of the UCL of surface water concentrations to represent exposure concentrations may overestimate risks to fish populations.

4.7.3.1.2 Surface Water Sampling Methods

Sampling methods used for the collection of surface water were either peristaltic or XAD or both. 2,4′-DDT and total DDTs exceed the chronic Eco SLs (HQs > 1.0) and are based on the surface water results from the peristaltic method. In addition, 2,4′-DDD, 4,4′-DDD, 4,4′-DDT and total DDTs exceed the chronic Eco SLs (HQs >1.0) and are based on the surface water results from the XAD high-volume method. These XAD HQs are greater than the peristaltic HQs. Some uncertainty may be associated with the data analyzed from the peristaltic samples because these samples were collected over a short period of time and may overestimate or underestimate exposure concentrations for and risks to fish.

4.7.3.1.3 Use of NJ-Qualified Data

UCLs for 2,4′-DDT and total DDTs based on the peristaltic method include NJ-qualified results. Results qualified as NJ indicates "the presence of an analyte that has been 'tentatively identified,' and the associated numerical value represents its approximate concentration" (EPA 1999). The qualification indicates that the analyst believed that the result was due to analytical interference from a chemical other than the target analyte. The NJ-qualified result is biased high and can result in an overestimation of risk.

For 2,4′-DDT, only 1 of 52 surface water samples collected from the peristaltic method were detected and it is a NJ-qualified result. In addition, the NJ qualified result (0.0187 µg/L) was approximately two orders of magnitude higher than any of the other results. The UCL based on the NJ-qualified result (0.0021 µg/L) results in a

chronic HQ of 2.1 (Table 4-52). However, the UCL excluding the NJ-qualified result is 0.00025 µg/L and results in a chronic HQ of 0.25. Therefore, the HQ for 2,4′-DDT based on peristaltic method likely overestimates risk to fish.

For total DDTs, only 4 of 52 surface water samples were detected and the maximum detected sample is a NJ-qualified result. The UCL based on the NJ-qualified result (0.0024 µg/L) results in a chronic HQ of 2.4 (Table 4-52). However, the UCL excluding the NJ-qualified result is 0.00063 µg/L and results in a chronic HQ of 0.63. Therefore, the HQ for total DDT based on peristaltic method likely overestimates risk to fish.

4.7.3.2 Effects Assessment

The appropriateness of the Eco SLs was evaluated as the primary uncertainty associated with the effects assessment for the surface water LOE. Eco SLs are based AWQC or ODEQ benchmarks that were developed based on fish or invertebrate species sensitivities and are considered protective of all aquatic receptors, including fish.

The Eco SL for 4-chloro-3-methyl phenol has some uncertainty associated with its use in assessing risks to fish. The chronic Eco SL for 4-chloro-3-methylphenol was calculated from an acute ODEQ value using a UF of 50, as directed by EPA (EPA 2006d). This chronic Eco SL may overestimate toxicity to fish receptors.

4.7.4 Transition Zone Water Assessment

The exposure of sculpin and lamprey ammocoetes to chemicals in TZW is uncertain. The ecological CSM (Figure 4-1) for all fish receptors shows TZW as an incomplete pathway; however, per EPA's request (EPA 2006a), TZW was evaluated as an exposure pathway to sculpin and lamprey ammocoetes. The ecological relevance of TZW to sculpin and lamprey ammocoetes is low. A porewater ventilation rate of 0 to 10% was assumed for sculpin and benthic invertebrates in the FWM (see Appendix E), and the same porewater ventilation rate was assumed for lamprey ammocoetes.

In addition, the TZW assessment assumes that organisms are exposed to single point locations of TZW areas; however, fish are mobile organisms and the assumption that sculpin and lamprey ammocoetes are exposed to TZW from a single point location is highly conservative and likely overestimates exposure to these fish receptors at the population level.

4.7.5 Summary of Uncertainties

The associated uncertainties for all Round 2 COPCs are presented in Tables 4-58 through 4-63 for each receptor evaluated. These uncertainties were used along with the results of the HQ calculations (Section 4.6.1) to make risk conclusions and identify Round 2 iCOCs for fish receptors.

4.8 RISK CONCLUSIONS

The results of the HQs presented in Section 4.6.1 were combined with the uncertainties associated with each Round 2 COPC/receptor presented in Section 4.6.2 to arrive at risk conclusions. The selected iCOCs have been identified for this Round 2 ERA and final COCs will be identified as part of the BERA. Round 2 initial chemicals of concern (iCOCs) were identified based on the following:

- For the tissue LOE, fish iCOCs were identified as those Round 2 COPCs that exceeded the LOAEL TRVs based on the UCL EPC (for wide-ranging fish receptors) or any individual composite location EPC (for small ranging receptors including sculpin, smallmouth bass, or northern pikeminnow), except for physiologically regulated metals, which were based on the dietary dose LOE.
- For the dietary dose LOE, fish iCOCs were identified as those Round 2 COPCs, including physiologically regulated metals, that exceeded the LOAEL TRV.
- For the surface water LOE, fish iCOCs were identified as those Round 2 COPCs for which the UCL exceeded Eco SLs and were supported by other LOEs as a chemical posing potential risk to fish.
- For the TZW LOE, fish iCOCs were those chemicals that were identified as potential iCOCs through the TZW framework and supported by other LOEs as chemicals posing potential risk to fish.

Based on the above rationale, the following chemicals were identified as iCOCs for fish.

PCBs – Total PCBs were identified as an iCOC for fish based on the dietary dose and tissue LOEs. In tissue, the total PCB LOAEL HQ exceeded 1.0 for smallmouth bass because a single composite at one location (08R010) exceeded a LOAEL TRV (HQ = 1.2). LOAEL HQs were less than 1.0 for all other fish receptors based on the tissue-residue LOE. Dietary dose LOAEL HQs exceeded 1.0 for all receptors except peamouth and juvenile chinook salmon (LOAEL HQs ranged from 0.33 to 1.8 for all fish receptors). Analysis of the surface water LOE resulted in LOAEL HQs less than 1.0.

DDTs – DDTs (including 2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDT, and 4,4′-DDE, and total DDTs) were identified as an iCOC for fish based on the tissue residue and TZW LOEs. In tissue, the total DDTs LOAEL HQ exceeded 1.0 for in a single sculpin tissue composite at one location (07R006). DDTs tissue-residue LOAEL HQs were less than 1.0 for all other fish receptors. Analysis of the dietary dose and surface water LOEs resulted in LOAEL HQs less than 1.0 for all receptors. Analysis of the

TZW LOE resulted in LOAEL HQ exceedances in a few locations but exceedances were not widespread.

BEHP – BEHP was identified as an iCOC for fish based on the empirical and predicted tissue-reside LOE. Sculpin and smallmouth bass LOAEL HQs based on UCL and at individual composite locations were > 1.0 (maximum HQ of 8.2). The predicted sculpin tissue concentration (based on the site-wide UCL sediment concentration) resulted in a LOAEL HQ of 6.7. The LOAEL HQs was less than 1.0 for largescale sucker and BEHP screened out as a tissue-residue Round 2 COPC for all other fish receptors. There is moderate uncertainty associated with BEHP as an iCOC based on the tissue LOE because detection frequency in sculpin and smallmouth bass tissue was low. Additionally, the BEHP tissue-residue LOAEL TRV is uncertain because limited toxicological studies were available and the TRVs were derived using an egg-to-adult conversion factor derived from PCB data. BEHP was not a dietary COI and was not a COPC for the surface water LOEs.

TBT – TBT was identified an iCOC for fish based on the dietary dose LOE. Dietary LOAEL HQs exceeded 1.0 for pre-breeding sturgeon, juvenile chinook salmon, smallmouth bass, and northern pikeminnow (LOAEL HQs = 1.1 for these receptors). However, there is very high uncertainty in the dietary LOE because only one study evaluating dietary effects of TBT on fish was available. Uncertainty analysis of the tissue-residue LOE showed that LOAEL HQs just slightly greater than 1.0 were moderately likely assuming invertebrate tissue data were representative of fish tissue data. Risk estimates for fish based on TBT may be improved with additional receptor tissue data (to evaluate TBT using the tissue-residue approach), however, the tissue-residue TRVs are also uncertain. TBT was not a COPC for the surface water LOE.

Mercury – Mercury was identified as an iCOC for fish based on the tissue-residue LOE. The northern pikeminnow tissue LOAEL HQ in a single composite at one location had a LOAEL HQ of 1.1, but the LOAEL HQ based on the tissue UCL was less than 1.0. Mercury was not a tissue COPC for all other fish receptors and the magnitude of exceedance at the single northern pikeminnow composite location was low; therefore, the iCOC designation of mercury is highly uncertain. Analysis of the dietary LOE resulted in LOAEL HQs less than 1.0 for pre-breeding sturgeon and northern pikeminnow. Mercury was not identified as a Round 2 COPC for any other fish receptors or for the water LOEs.

Cyanide, perchlorate, and 16 individual PAHs (i.e., 2-methylnaphthalene, acenaphthene, anthracene, benzo[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene, benzo[g,h,i]perylene, benzo[k]fluoranthene, chrysene, dibenzo[a,h]anthracene, fluoranthene, fluorene, indeno[1,2,3-cd]pyrene, naphthalene, phenanthrene, and pyrene) were retained as potential iCOCs for fish. Potential iCOCs for fish were identified as those chemicals identified by the TZW LOE, but not supported by other LOEs, as posing risks to fish. These potential iCOCs are not

expected to pose significant risks to fish because there is limited potential for exposure.

Two additional Round 2 COPCs (i.e., cadmium and copper) for which the uncertainty analysis resulted in a high or moderate likelihood that the LOAEL HQ would exceed 1.0 were not identified as iCOCs for fish; however, these Round 2 COPCs will be included in the BERA to confirm that these chemicals do not pose a risk. Dietary dose LOAEL HQs for cadmium were all < 1.0 and cadmium was not identified as a Round 2 COPC in any other LOE. However, based on uncertainty analyses, there is a moderate likelihood that a plausible change in the sediment ingestion rate would result in a LOAEL HQ greater than 1.0 for smallmouth bass and a moderate likelihood that reducing the exposure scale would result in LOAEL HQs >1.0 in localized areas for sculpin. Similarly, Dietary dose LOAEL HQs for copper were all < 1.0 and copper was not identified as a Round 2 COPC in any other LOE. An evaluation of copper concentrations on migratory behavior was conducted and concluded that concentrations in the Study Area are unlikely to cause a change in migrating behavior (see Attachment G4). However, based on uncertainty analyses, the likelihood that a plausible change in dietary exposure assumptions would result in LOAEL HQs greater than 1.0 is high pre-breeding sturgeon and a moderate for smallmouth bass. Additionally, there is a moderate likelihood that reducing the exposure scale would result in a LOAEL HQ greater than 1.0 for sculpin. Cadmium and copper were not COPCs for the tissue or surface water LOEs, and were not identified as a potential iCOC in TZW.

All other Round 2 COPCs were not identified as iCOCs or recommended to be carried forward to the BERA. Some Round 2 COPCs (i.e., lead and selenium) had LOAEL exceedances for at least one LOE but were excluded as iCOCs because the most ecologically relevant LOE for that COPC resulted in HQs less than 1.0. The peamouth LOAEL HQ exceeded 1.0 for lead, but lead was not retained as an iCOC for fish because lead screened out as a Round 2 COPC for the dietary dose LOE showed no risk to fish and the dietary dose LOE is a more ecologically relevant LOE for physiologically regulated metals. The predicted sculpin tissue concentration of selenium (predicted using a BSAF and the UCL site-wide sediment concentration) resulted in a LOAEL HQ of 1.1; however, because the magnitude of the LOAEL HQ was low and because the empirical tissue LOE and other LOEs do not indicate that selenium may pose a risk to fish, selenium was not retained as an iCOC for fish.

Fish iCOCs identified were mapped (presented in Section 10 of the Comprehensive Round 2 Report) to identify initial areas of potential concern (AOPCs) for fish receptors. Potential data needs for fish in the BERA are presented in Section 8.0.

5.0 WILDLIFE RISK ASSESSMENT

This section presents the Round 2 ERA for wildlife in the Portland Harbor Study Area. The problem formulation (Section 5.1) identifies the selected ecological wildlife receptors, exposure pathways, and lines of evidence evaluated for determining risks to wildlife species that use the LWR and the results of the screening assessment to identify wildlife Round 2 COPCs. Round 2 COPCs for each line of evidence were identified using data collected to date based on screening level evaluations for each line of evidence. Sections 5.2 and 5.3 present the exposure and effects methods for evaluating risks to wildlife receptors via the dietary dose and bird egg tissue risk assessment methods, respectively for all Round 2 COPCs identified for wildlife. Round 2 COPCs are carried forward in a risk evaluation to identify the iCOCs that will be further evaluated in the BERA. Section 5.4 presents the Round 2 risk conclusions and iCOCs that will inform the BERA.

5.1 PROBLEM FORMULATION

This section briefly presents the problem formulation for the wildlife receptors in the LWR. Details on the wildlife problem formulation are presented in Appendix B of the Programmatic Work Plan (Integral et al. 2004b). Some aspects of the problem formulation and the risk analysis process have evolved since the initial presentation in the Programmatic Work Plan. The following text reflects the changes that have been agreed upon with EPA and its partners.

5.1.1 Wildlife Resources Potentially at Risk

Section 2.0 of the Comprehensive Round 2 Report presents a general description of the environmental setting in the LWR. The intertidal and subtidal habitats of the LWR support a diversity of wildlife species. A summary of the habitat utilization of wildlife species and the potential wildlife receptors at risk were identified in Appendix B of the Programmatic Work Plan (Integral et al. 2004b). The bird and mammal receptors that use or may potentially use the LWR are discussed in the following subsections.

5.1.1.1 Birds

Numerous aquatic and semi-aquatic bird species use the Willamette River. Bird species that were identified as using habitat within the Study Area were grouped into the following feeding guilds:

- **Herbivores** – birds that feed predominately on plant material
- **Diving carnivores and omnivores** – birds that usually swim on the surface and feed on invertebrates or a mix of invertebrates, fish, and occasionally plants taken from the sediment surface

- **Sediment-probing invertivores and omnivores** – birds that probe in the sediments for invertebrates in shallow water along the shoreline
- **Piscivores** – birds that feed exclusively on fish

For all bird species, exposure in the Study Area is mainly due to ingestion of contaminants in food, or incidentally ingested sediment, soil, or water. Different habitats, prey species, and feeding behaviors result in different potential for exposure to sediment-borne contaminants. Exposure and risk was estimated for species that represent the major exposure pathways from each feeding guild.

Table 5-1 presents the aquatic and semi-aquatic bird species that may breed along the LWR. One bird species (i.e., bald eagle) identified as a potential resident bird to the LWR is a special-status (federally and state-threatened) species. Table 5-2 presents a list of bird species that may overwinter or only partially use the LWR. Appendix B of the Programmatic Work Plan (Integral et al. 2004b) provides additional information on feeding and habitat preferences and Willamette River site use of potential resident species within these feeding guilds. Five bird species (i.e., American peregrine falcon, Harlequin duck, red-necked grebe, black tern, and tri-colored blackbird) that were identified as species that seasonally or minimally associated with aquatic habitat in the Study Area are also special-status species (federally or state listed).

5.1.1.2 Mammals

Aquatic and semi-aquatic mammals that potentially use the LWR are presented in Table 5-3. Appendix B of the Programmatic Work Plan (Integral et al. 2004b) provides additional information on the feeding and habitat preferences of mammals that potentially use in the LWR. None of these mammal species are special-status species.

5.1.2 Selected Receptors of Concern

To select representative wildlife species, birds and mammals in the Study Area were evaluated using the systematic criteria presented in Appendix B of the Programmatic Work Plan (Integral et al. 2004b) and divided into primary feeding guilds: herbivorous birds, sediment-probing invertivorous and omnivorous birds, diving carnivorous and omnivorous birds, piscivorous birds, and carnivorous mammals. The following wildlife species were selected as ecological receptors to represent the various feeding guilds in the LWR:

- **Spotted sandpiper** – sediment-probing invertivorous and omnivorous birds (surrogate receptor for herbivorous birds)
- **Hooded merganser** – diving carnivorous and omnivorous birds
- **Bald eagle** – piscivorous birds
- **Osprey** – piscivorous birds

- **Mink** – carnivorous mammals
- **River otter** – carnivorous mammals

These selected receptor groups were presented in Appendix B of the Programmatic Work Plan (Integral et al. 2004b), which was conditionally approved by EPA and its partners in June 2004. The following subsections summarize the rationale for selecting the ecological receptor groups that will be assessed for the Portland Harbor Superfund Site ERA.

5.1.2.1 Spotted Sandpiper

Sediment-probing birds consume mostly sediment-associated invertebrates and may incidentally ingest more sediment than other birds. In the vicinity of the Study Area, sediment-probing birds include sandpiper, killdeer, sora, yellowlegs, marbled godwit, long-billed curlew, Wilson's phalarope, dowitcher, common snipe, and Virginia rail. Spotted sandpiper, killdeer, sora, Virginia rail, and common snipe are species that breed in the vicinity of the Study Area. None of the sediment-probing birds are listed as threatened or endangered, and species-specific laboratory toxicological data are not available for any of these species. Of the bird species evaluated for their consumption of soil and sediment by Beyer et al. (1994) sandpiper was found to have the highest sediment ingestion rate. Therefore, spotted sandpiper was chosen as a representative species based on sediment exposure and presence of breeding population. Because exposure of the spotted sandpiper to sediment is expected to be relatively high, the spotted sandpiper is considered to be a conservative representative of omnivorous and herbivorous birds.

Because of their diet and relatively high rate of incidental sediment ingestion, exposure of sediment-probing birds to sediment contaminants is higher than other groups such as herbivorous birds and dabbling ducks which feed on diets that are likely to have lower chemical concentrations and/or ingest substantially less sediment. Consequently, the spotted sandpiper provides a protective risk scenario on which to assess risk, and risk management decision made to protect sandpipers would be protective of these other guilds (e.g., herbivorous birds). Therefore, the exposure analysis did not include herbivores and dabbling ducks.

5.1.2.2 Hooded Merganser

Diving birds in the Study Area that ingest primarily invertebrates and small fish include bufflehead, hooded merganser, goldeneye, grebe, Bonaparte's gull, California gull, and scaup. None of these species are identified as threatened or endangered. Any of these species would be a good representative species for diving birds. Two diving carnivores (i.e., Harlequin duck and red-necked grebe) are special-status species; however, these species are transients in the LWR and do not mate or rear young in the LWR. The hooded merganser was selected as the representative species because it breeds and rears young in the area, and uses most of the aquatic habitat in the LWR to forage for crustaceans, mollusks, aquatic insects, and fish.

5.1.2.3 Bald Eagle and Osprey

Piscivorous birds that feed from the Willamette River in the vicinity of the Study Area include osprey, cormorant, heron, tern, eagle, kingfisher, American white pelican, and western gull. Of these birds, two species, the osprey and the bald eagle, were selected as representative species for piscivores, as agreed upon by LWG, EPA, and EPA's partners and presented in Appendix B of the Programmatic Work Plan (Integral et al. 2004b). The bald eagle is listed as threatened under the Endangered Species Act (ESA) and is also protected by the Bald and Golden Eagle Protection Act (16 USC 668), and both the osprey and bald eagle are protected under the Migratory Bird Treaty Act (16 USC 703-712). Osprey and bald eagle nests have been observed in or close to the Study Area, indicating that sensitive, developmental life stages of these species are potentially exposed to chemicals in the Study Area. Most of the osprey's prey is fish, and bald eagles are assumed to consume primarily fish in the LWR. Consumption of secondary aquatic consumers, such as invertivorous fish, gives bald eagle and osprey the highest potential exposure to biomagnifying chemicals. Chemical exposure in the Study Area for these two species is likely to be different because the bald eagle is present year-round and osprey is a migratory species, present only from spring through fall.

Other piscivores considered for inclusion as representative species were the cormorant, great blue heron, and kingfisher, based on their presence in the Study Area, breeding habitat, special regulatory status, or societal value. The great blue heron nests on Ross Island, but their site affiliation may not be as strong as that for the osprey because great blue heron may feed on upland species. Osprey and bald eagle both consume secondary consumers and have greater societal value than other piscivorous birds, and are likely the species most highly exposed to site-related chemicals. Therefore, osprey and bald eagle were selected receptors to representative piscivorous birds.

5.1.2.3 Mink and River Otter

The key exposure route for aquatic or semi-aquatic mammals is likely to be the consumption of prey associated with sediment in the Study Area. The mammalian species that may use the Study Area and have the greatest percentage of aquatic prey in their diets are mink, river otter, raccoon, and California sea lion. None of these mammal species are listed as threatened, endangered, or sensitive; however, the California sea lion is protected under the Marine Mammals Act. Raccoons are omnivores that ingest significant amounts of vegetation along with a broad range of other food items, and may obtain a significant proportion of their food from sources other than the LWR. The California sea lion is occasionally present in the LWR, and usually feeds on in-migrating adult salmon at Willamette Falls and the fish ladder. Since adult salmon spend the great majority of their lives outside of the Study Area, risk to sea lions from salmon is not considered a good indication of risk from the Study Area.

Mink were selected as one representative species for wildlife that consume aquatic prey. Mink are not common in the Study Area. However, mink are known to be extremely sensitive to PCBs and other chemicals, and therefore may represent extremely sensitive species of aquatic feeding mammals. The river otter may be less sensitive to contaminants than mink, and may be representative of a broader range of species. Therefore, as agreed to by LWG, EPA, and EPA's partners, the river otter was also evaluated as a selected mammal receptor.

5.1.3 Exposure Pathways

Potential chemical exposure pathways for the representative wildlife receptors in Portland Harbor were identified in Appendix B of the Programmatic Work Plan (Integral et al. 2004b). Receptors may be exposed to chemicals in water or sediment in Portland Harbor, either directly through contact with or ingestion of sediments or surface water or indirectly through the food chain. The conceptual site model (CSM) was developed for the Programmatic Work Plan (Integral et al. 2004b) and is presented in Figure 5-1. The CSM illustrates the pathways that chemicals may follow from primary sources to the ecological receptors.

Surface water exposure by direct contact (swimming) and ingestion (as drinking water) are considered complete and minor for all wildlife receptors. TZW exposure by direct contact (foraging for prey) and ingestion (as drinking water) are considered incomplete pathways for all wildlife receptors. As stated in Appendix B of the Programmatic Work Plan (Integral et al. 2004b), these pathways were not evaluated quantitatively in the risk assessment for wildlife as separate lines of evidence because water ingestion does not contribute measurably to total exposure for any contaminant when included in the calculation of risk estimates for birds and mammals, and uptake through direct (dermal) contact with water is limited and assumed to be minor because feathers (on birds) and fur (on mammals) insulate skin and limit the direct contact of skin with water. Furthermore, uptake through exposure of birds and mammals to surface water, seep water, and TZW is considered minor because of the relative insolubility of many COIs.

5.1.3.1 Spotted Sandpiper

Spotted sandpipers represent birds that primarily consume benthic invertebrates; thus, ingestion of benthic biota is considered a complete and major pathway of exposure. In addition, spotted sandpipers are likely to consume sediment during feeding. Thus, sediment ingestion is considered a complete and major pathway of exposure. Both the ingestion of benthic biota and surface sediment were evaluated quantitatively in the dietary dose line of evidence (Section 5.2).

Because spotted sandpipers forage extensively on beaches, they are likely to be in direct contact with sediments. Direct contact exposure studies have not been conducted with shorebirds; therefore, the contribution to total exposure from direct contact is uncertain. Although assumed to be a complete pathway, this pathway was not quantitatively assessed because there is insufficient toxicological data. Risks from

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sediment contact are considered to be insignificant relative to those from ingestion (EPA 2000b).

Surface water direct contact and ingestion are considered complete but relatively minor pathways and were not evaluated quantitatively. The impact of surface water chemical exposure via ingestion on the overall exposure dose estimates for spotted sandpiper is discussed in the uncertainty section (Section 5.4.2).

5.1.3.2 Hooded Merganser

Hooded mergansers represent birds that primarily consume benthic invertebrates and small fish. Therefore, the ingestion of biota is considered a complete and major pathway of exposure. In addition, hooded mergansers are likely to ingest sediment incidentally while foraging and indirectly through their prey. Thus, sediment ingestion is considered a complete and major pathway. Both the ingestion of biota and surface sediment were evaluated quantitatively in the dietary dose line of evidence (Section 5.2).

Surface water ingestion and direct contact with surface water and sediment are considered incidental occurrences and complete but relatively minor pathways of exposure, and these pathways were not quantitatively evaluated. The impact of surface water chemical exposure via ingestion on the overall exposure dose estimates for hooded merganser is discussed in the uncertainty section (Section 5.4.2).

5.1.3.3 Bald Eagle and Osprey

Piscivorous birds, such as osprey and bald eagle, primarily consume fish from the water column. Therefore, the ingestion of fish and other biota is considered a complete and major pathway of exposure for these species. Bald eagles frequently consume beached fish carcasses and may incidentally consume sediment as a result. Thus, sediment ingestion is considered a complete and minor pathway of exposure for bald eagle and osprey. Both the ingestion of biota and surface sediment were evaluated quantitatively in the dietary dose line of evidence (Section 5.2) and the dietary model for estimating bird egg concentrations for both bald eagles and osprey (Section 5.3).

Inasmuch as surface water contact, surface water ingestion, and direct sediment contact are likely to be minimal, these pathways are considered to be complete but minor pathways and were not quantitatively evaluated. The impact of surface water chemical exposure via ingestion on the overall exposure dose estimates for bald eagle and osprey is discussed in the uncertainty section (Section 5.4.2).

5.1.3.4 Mink and River Otter

Carnivorous mammals consume a wide range of prey but primarily consume aquatic prey such as fish and macroinvertebrates (e.g., crayfish); therefore, the ingestion of fish and other biota is considered a complete and major pathway of exposure. Mink and river otter may ingest sediment while foraging, burrowing, or digging into the sediment and while grooming their fur. Thus, sediment ingestion is considered a

complete and major pathway of exposure for carnivorous mammals. Both the ingestion of biota and surface sediment were evaluated quantitatively in the dietary dose line of evidence for mink and river otter (Section 5.4.2).

Surface water ingestion and direct contact with surface water and sediment are considered complete but relatively minor pathways, and these pathways were not quantitatively evaluated. The impact of surface water chemical exposure via ingestion on overall exposure dose estimates for mink and river otter is discussed in the uncertainty section (Section 5.4.2).

Direct sediment contact may contribute to total exposure through dermal absorption. However, given the protective properties of the fur of mink and other aquatic carnivorous mammals that may use the Study Area, direct contact would be limited to the paws (and possibly to the nose, and/or eyes), and risks from sediment contact are considered to be insignificant relative to those from ingestion (EPA 2000b). Thus, this pathway is considered a complete and minor pathway and was not quantitatively assessed.

5.1.4 Lines of Evidence Approach and Methods

Lines of evidence for assessing risks to wildlife were presented in Appendix B of the Programmatic Work Plan (Integral et al. 2004c) and were further refined based on discussions between LWG, EPA, and EPA's partners (EPA 2004a, 2005a). LWG agreed to evaluate all of the lines of evidence for assessing risks to wildlife receptors for the Comprehensive Round 2 Report. Table 5-4 presents the lines of evidence evaluated for wildlife receptors. Direct measurement data at the population , if available.

The weighting scheme for the LOEs for wildlife depended on the purpose of the assessment/task, of which there were four:

- Screening COIs to identify Round 2 COPCs
- Further screening of Round 2 COPCs to identify iCOCs
- Identifying potential data needs for the BERA
- Providing information that would be used to identify iAOPCs

Round 2 COPCs were identified in the SLERA. The other three tasks were done to better prepare for the BERA and other parts of the RI/FS that may use risk assessment results.

The weight-of-evidence approach for Round 2 COPC identification was as follows:

- Round 2 COPCs were identified for each LOE.
- The Round 2 COPC lists were integrated across LOEs to derive the overall list of Round 2 COPCs for fish.

This is a conservative approach in that any chemical that screened in based on any one LOE was carried forward as a Round 2 COPC.

The identification of iCOCs, potential data needs, and iAOPCs used the same weight-of-evidence approach as that used for Round 2 COPC identification, differing in level of conservatism. The identification of Round 2 COPCs used an appropriate level of conservatism for a SLERA. The iCOC identification used less-conservative assumptions and incorporated a weighted approach across LOEs in an effort to narrow the list of chemicals and identify the potential areas associated with risks to wildlife in the Study Area. This was important for identifying data needs and for developing iAOPCs that anticipate where the AOPCs (to be developed in the BERA) will be.

Wildlife receptors were evaluated using two types of dietary models. The first model was used to assess risks to all wildlife receptors by estimating dietary exposure doses through prey tissue and incidental sediment ingestion and receptor-specific parameters. These dietary doses were compared to dietary TRVs derived from the literature. The second dietary model was developed for piscivorous birds (i.e., bald eagle and osprey) based on the availability of TRVs expressed as Round 2 COPC concentrations in eggs. Bird egg tissue concentrations for bald eagle and osprey exposed within the Study Area were estimated based on fish prey tissue concentrations and biomagnification factors (BMFs). Estimated bird egg tissue concentrations were compared to bird egg TRVs. Data collected from the Study Area were considered representative of the prey items in each of the wildlife receptor's diets. Thus, empirical prey tissue concentrations were used to estimate exposure in both dietary models.

Because surface water was expected to be a minor pathway of exposure for wildlife receptors, surface water ingestion was evaluated and is presented in the uncertainty section (Section 5.4).

5.1.5 Screening Summary for Round 2 COPC Identification

A screening step was conducted for each line of evidence presented in Table 5-4 to identify Round 2 COPCs for wildlife risk evaluation. Attachment G6 presents details on the screening process and identification of Round 2 COPCs for the dietary dose and bird egg tissue lines of evidence, and a summary of the results are presented in the following subsections.

5.1.5.1 Dietary Dose Screen and Round 2 COPC Identification

To identify Round 2 COPCs for wildlife, the maximum exposure dose (IR_{diet}) of COI was compared to a NOAEL TRV for that chemical. If the maximum exposure dose (for each receptor) was greater than the NOAEL TRV, the chemical was identified as a Round 2 COPC. The identification of COPCs through a conservative screen in which maximum concentrations in relevant media are used is consistent with EPA

guidance (EPA 1997b, a) and an important step for narrowing the list of chemicals to only those that could potentially pose a risk to ecological receptors.

A summary of the Round 2 COPCs that were evaluated further for each wildlife receptor are presented in Table 5-5 for each wildlife receptor.

5.1.5.2 Tissue-Residue Screen and Round 2 COPC Identification

To identify Round 2 COPCs for using the bird egg approach, the maximum exposure egg concentration (EPC_{egg}) of each COI was compared to a NOAEL TRV for that chemical. If the maximum exposure dose (for both osprey and bald eagle) was greater than the NOAEL, the chemical was identified as a Round 2 COPC.

Estimates of the all five bird egg COIs (dioxin TEQ, PCB TEQ, total PCBs, DDE, and mercury) were greater than their respective NOAEL TRVs. These chemicals were retained as Round 2 COPCs for osprey and bald eagles and were evaluated in Section 5.3.2.

5.2 DIETARY DOSE ASSESSMENT

The primary line of evidence for evaluating risks to all wildlife receptors was the dietary dose assessment. Daily dietary doses were quantitatively estimated for each chemical and wildlife receptor and included the ingestion of biota (i.e., prey) and incidental ingestion of sediment. Other pathways considered in the CSM (Figure 5-1) were determined to be minor relative to these primary exposure pathways and not important for risk management decisions. Wildlife exposure to ingested surface water was considered a minor pathway; the evaluation of this assumption is discussed in the uncertainty section (Section 5.4.2).

Dietary doses were estimated using the following equation:

$$IR_{diet} = \frac{\left[\left(FIR \times C_{prey}\right) + \left(SIR \times C_{sed}\right)\right]}{BW} \times SUF \qquad \textbf{Equation 5-1}$$

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Where:

IR_{diet}	=	estimated bird or mammal exposure dose or intake rate (mg/kg bw/day)
FIR	=	food ingestion rate (kg ww food/day)
C_{prey}	=	tissue concentration in prey items (mg/kg ww)
SIR	=	sediment ingestion rate (kg dw sediment/day)
C_{sed}	=	concentration in surface sediment (mg/kg dw)
BW	=	body weight (kg)
SUF	=	site use factor (unitless); fraction of time that a receptor spends foraging in the Study Area relative to their entire home range

Body weights, food ingestion rates (FIRs), sediment ingestion rates (SIRs), and site use factors (SUFs) vary among wildlife receptors.

5.2.1 Round 2 COPC Exposure Assessment

This section presents the exposure assessment for those wildlife Round 2 COPC/receptor pairs identified in Table 5-5 (see Section 5.1.5).

5.2.1.1 Approach

Round 2 COPC exposure estimates were expressed as a dietary dose, representing prey tissue and incidental sediment ingestion. The daily doses were estimated using Equation 5-1:

$$IR_{diet} = \frac{\left[\left(FIR \times C_{prey}\right) + \left(SIR \times C_{sed}\right)\right]}{BW} \times SUF \qquad \textbf{Equation 5-1}$$

Where:

IR_{diet}	=	estimated bird or mammal exposure dose or intake rate (mg/kg bw/day)
FIR	=	food ingestion rate (kg ww food/day)
C_{prey}	=	tissue concentration in prey items (mg/kg ww)
SIR	=	sediment ingestion rate (kg dw sediment/day)
C_{sed}	=	concentration in surface sediment (mg/kg dw)
BW	=	body weight (kg)
SUF	=	site use factor (unitless); fraction of time that a receptor spends foraging in the Study Area relative to the entire home range

Body weights, food ingestion rates, sediment ingestion rates, and site use factors vary among wildlife receptors. Table 5-6 presents the dietary assumptions that were used to estimate dietary exposure doses for birds and mammals. In accordance with EPA comments (EPA 2006b), dietary doses for all wildlife receptors were based on female exposure parameters.

The following subsections describe how these parameters were selected for each bird or mammal receptor species (Sections 5.2.2.2 through 5.2.2.7). Body weights and

food ingestion rates were primarily based on the literature presented in EPA's *Wildlife Exposure Factors Handbook* (EPA 1993). When species specific data were not available in EPA (1993), food ingestion rates were based on the allometric equations presented in Nagy (2001). SIRs were derived based on Beyer et al. (1994), or on best professional judgment when no data were available. Site use factors were based on species specific data from regional studies.

All Round 2 COPCs were conservatively assumed to have the same bioavailability in the field as in the laboratory toxicity study that provides the basis for the TRV in all media. C_{prey} and C_{sed} were estimated from Study Area data using appropriate EPCs in relevant prey tissue items and surface sediment from relevant exposure areas. EPCs were calculated for all Round 2 COPCs in surface sediment and for each of the prey tissue items as an upper bound estimate (UCL of the mean). Details on the calculation of UCLs are presented in Attachment 1 and EPCs of C_{prey} and C_{sed} for each wildlife receptor are presented in Section 5.2.2.8.

The chemical concentration in each receptor's prey (C_{prey}) was calculated from the UCL concentrations in each component of the receptor's diet and estimates of each component's fraction of the total diet. For example, the concentrations in food for a receptor that might ingest both fish and benthic invertebrate prey would be estimated as follows:

$$C_{prey} = (C_f \times F_f) + (C_b \times F_b)$$ **Equation 5-3**

Where:

C_{prey}	=	concentration in prey items (mg/kg ww)
C_f	=	UCL concentration in fish prey tissue (mg/kg ww)
F_f	=	fraction of the receptor diet consisting of fish (kg fish/kg food)
C_b	=	UCL concentration in benthic invertebrates prey tissue (mg/kg ww)
F_b	=	fraction of the receptor diet consisting of benthic invertebrates (kg benthic invertebrates/kg food)

Table 5-7 presents the prey species available from the ERA dataset (Section 2) that were used to represent wildlife prey items and the portion of the wildlife diet that each prey item comprises to calculate the concentration in prey (C_{prey}; see Equations 5-1 and 5-3). The dietary fraction of each component in each receptor's diet was based on information from the literature. Prey items included fish and invertebrate species (i.e., largescale sucker, carp, juvenile chinook salmon, sculpin, peamouth, smallmouth bass, northern pikeminnow, black crappie, and brown bullhead, crayfish, clams) collected in the field within the Study Area and invertebrate species (i.e., worms) that underwent bioaccumulation testing. Clams collected in the field were used to represent clam prey for the wildlife receptors; clams that underwent bioaccumulation testing were not used in the dietary prey scenarios.

Not all Round 2 COPCs were analyzed in all prey items. Therefore, surrogate species were selected from similar feeding guilds to represent prey tissue concentrations. These surrogate species for specific Round 2 COPCs are noted in Table 5-7.

The following subsections detail the rationale for each of the assumptions made for each wildlife receptor to estimate dietary exposure doses.

5.2.1.2 Dietary Exposure Assumptions

This section describes how the exposure parameters were selected for each wildlife receptor species (i.e., spotted sandpiper, hooded merganser, bald eagle, osprey, mink, and river otter), as presented in Tables 5-6 and 5-7.

5.2.1.2.1 Spotted Sandpiper

Body Weight and Daily Food Ingestion Rate

Maxson and Oring (1980), as presented in EPA (1993), reported average adult female and male body weights to be 0.0471 and 0.0379 kg, respectively. Daily food ingestion rates were estimated as a function of body weight derived from Nagy (2001), who reported the body weight-normalized daily food ingestion rates for common sandpiper as 0.175 g dw/g bw/day. Using the common sandpiper ingestion rate and the female body weight for spotted sandpiper, the calculated food ingestion rate for spotted sandpiper was 0.0082 kg dw/day. The food ingestion rate was converted in to wet weight using a percent moisture of 85%, resulting in female spotted sandpiper food ingestion rate of 0.0548 kg ww/day. Eight-five percent moisture was the average percent moisture in shorebird prey items (invertebrate tissue) collected from the Study Area.

Incidental Sediment Ingestion Rate

Sandpipers are known to ingest large amounts of sediment while feeding on benthic prey. Beyer et al. (1994) reported that sediment ingestion by four sandpiper species (i.e., stilt sandpiper, semipalmated sandpiper, least sandpiper, and western sandpiper) ranged from 7.3 to 30% of the dry diet, with an average incidental sediment ingestion of 18%. The female dietary SIR of 0.00148 kg dw/day was calculated assuming 18% incidental sediment ingestion.

Diet Composition

Spotted sandpipers feed primarily on terrestrial and aquatic insects (Bent 1929; Csuti et al. 2001). They may occasionally feed on other benthic macroinvertebrates such as crustaceans, mollusks, and worms (Bent 1929; Csuti et al. 2001) or on leeches, small fish, and carrion (Oring et al. 1983). Amphipod or insect tissue was not available from the LWG Study Area; however, field clams were used as a surrogate species to represent the diet of spotted sandpiper. The diet was modeled assuming 100% prey ingestion of field clams from within the Study Area.

An additional dietary scenario was evaluated for sandpipers. In dietary scenario two, sandpipers were assumed to consume worms as 100% of their prey. Worm tissue from the bioaccumulation testing (based on surface sediment collected from within

the Study Area) were used to evaluate risk to sandpiper. The uncertainty of using bioaccumulative tissues to estimate exposure to shorebirds foraging in the field is discussed in the uncertainty section (Section 5.4.2).

Site Use and Exposure Area

A site use factor of 1.0 was used to estimate exposure of spotted sandpipers because they are locally common breeders in the Willamette Valley, and some are present year-round (Puchy and Marshall 1993).

Potential shorebird foraging beach habitats were identified as part of the Round 2 shorebird beach sampling field collection. LWG and EPA identified and agreed upon shorebird beach areas during the reconnaissance for the Round 2 shorebird sampling described in the July 16, 2004 reconnaissance memo from LWG to EPA (Saban and Andersen 2004). These beach areas, in conjunction with the human use beach areas collected during Round 1 sampling were used to evaluate risks to shorebirds.

Figure 5-2 presents the 28 shorebird beach locations (B1 through B28) that were assessed individually for shorebird exposure. Surface sediment represented by random samples collected along transects representing each beach area was collected during Round 1 or Round 2 (Integral et al. 2004d; Integral 2005). These sediment samples were used to assess exposure of shorebirds to sediments at each beach location. TBT was not analyzed in any shorebird beach sediment and TEQ analysis in beach sediment was limited to a subset of beach samples. Therefore, TBT, PCB TEQ, and dioxin TEQ exposure at each beach was evaluated using the co-located sediment collected with Round 2 clam and bioaccumulation sediment, where no transect beach sediment data were available.

Tissue data (from both field-collected clam and worms that underwent bioaccumulation testing from co-located sediment) were identified from within each beach area or adjacent to each beach area to represent dietary exposure of shorebirds to each beach area. Table 5-8 presents the samples associated with each beach area and the approximate area of each beach. For those beaches where insufficient field clam data was collected to analyze all Round 2 COPCs (i.e., B10, B11, B19, B21, B24, B27, and B28), laboratory clam data were used to represent clam tissue for those Round 2 COPCs where no field clam tissue data was analyzed.

The ecological relevance of assessing risk to shorebirds at each individual foraging beach was evaluated using foraging and dispersal distances reported in the literature. Western sandpipers were reported to move 4 to 6 km (2.5 to 3.7 miles) while foraging on beaches during migratory stopover in the Fraser River Estuary in British Columbia (Butler et al. 2002). Similarly, western sandpipers in San Francisco Bay have been noted to use a relatively small home range area (22 km^2 [22,000,000 m^2] or 8.5 square miles) during their winter and spring base stopover (Warnock and Takekawa 1995). Reed and Oring (1993) reported that dispersal distances of post-fledging hatch year and adult birds range from 22.8 to 146.7 km (14 to 91 miles) in Minnesota over an eight-year study. Breeding territory sizes tend to be quite small, ranging from 812 –

20,000 m^2 with beach length from 20 to 400 m long (Oring et al. 1997). The approximate area of each shorebird beach ranged from 4,656 to 188,665 m^2. Fifteen of these beach areas are within the size of breeding territories (812 to 20,000 m^2) and all are less than the home range areas or foraging distances reported in the literature. The assumption that each beach area could represent breeding habitat for shorebirds is highly conservative; it assumes that breeding would occur at each of these beach areas and that shorebirds would forage only from within that designated beach area and not from adjacent beaches. The uncertainty of assessing risks to shorebirds at each individual beach habitat areas discussed in the uncertainty section (Section 5.4.2).

Each shorebird beach area was given a qualitative "high" or "low" habitat quality designation (Table 5-8). This habitat designation is qualitative and was assigned to each beach area to evaluate risk conclusions and the likelihood of shorebird populations in each habitat beach area:

- High quality habitat beaches were considered to have moderate to high shorebird foraging habitat. High quality beaches had continuous open stretches of sand or mud substrate with some debris present in the open intertidal area. Upland vegetation was identified adjacent to high quality habitat beaches and beaches had a slight to moderate gradient slope.
- Low quality habitat beaches were considered to have limited foraging habitat. These beaches had limited open areas or stretches of sandy beach (even during low water conditions), had a significant presence of large rocks, riprap, or gravel, and/or had a steep slope.

In addition to evaluating shorebird exposure at each individual beach location, all beaches were evaluated together (site-wide) to evaluate the potential exposure of shorebird populations utilizing the entire 9 miles of the Study Area. In this evaluation, the UCL of the sediment from the all beach areas (B1 through B28) and the UCL of the clam and worm tissue from throughout the Study Area were used to calculate shorebird exposure to populations utilizing beach habitat throughout the Study Area.

No worm or clam tissue was collected within or adjacent to five shorebird beach areas (i.e., B1, B4, B7, B8, and B14). For these five beaches, worm and clam tissue concentrations were represented using one of three methods:

- Use of the food web model (FWM) to predict tissue concentrations in worms and clams. Concentrations of worm and clam tissue were estimated using the FWM (Appendix E) for the following Round 2 COPCs: dioxin TEQ, PCB TEQ, total PCBs, sum DDD, sum DDE, sum DDT, total DDTs, and

aldrin[47] in those beaches where worm and/or field clam data were not available. The uncertainty associated with the use of the FWM to predict benthic tissue concentrations is discussed in the uncertainty section (Section 5.4.2).

- Use of a site-specific and species-specific regression to predict tissue concentrations in worms and clams. Concentrations of worm and/or clam tissue were estimated using a regression for Round 2 COPCs not modeled via the FWM and for those Round 2 COPCs for which a relationship between sediment and tissue was found (Appendix E). For clams, BSAFs were used to estimate tissues at beaches with no clams tissue data for cadmium, lead, benzo(a)pyrene, and total PAHs. For worms, regression were used to estimate tissues at beaches with no worm tissue data for the lead, zinc, and benzo(a)pyrene. Table 5-9 presents the site-specific regressions that were used to estimate worm and clam tissue concentrations.
- Use of the UCL of all worm and clam tissue collected in the Study Area to represent worm and clam tissue concentrations. For those non-FWM Round 2 COPCs for which no site-specific regression was found, a UCL of the prey tissue (worm and/or clam) was used to represent tissue concentrations at beaches where no tissue were collected. For worms, a UCL was used to represent tissue at beaches with no worm tissue data for the following Round 2 COPCs: arsenic, cadmium, copper, mercury, selenium, total PAHs, BEHP, and dibutyl phthalate. For clams, a UCL was used to represent tissue at beaches with no clam tissue data for the following Round 2 COPCs: arsenic, copper, mercury, selenium, zinc, endrin, endrin aldehyde, endrin ketone, BEHP, and dibutyl phthalate.

For those beaches where insufficient field clam data were collected to analyze all Round 2 COPCs (i.e., B10, B11, B19, B21, B24, B27, and B28), laboratory clam data were used to represent clam tissue for those Round 2 COPCs where no field clam tissue data was analyzed. Table 5-9 presents the site-specific regressions used to estimate benthic tissue concentrations.

5.2.1.2.2 Hooded Merganser

Body Weight and Daily Food Ingestion Rate

Dunning (1993) reported adult female hooded merganser BWs ranging from 0.54 to 0.68 kg and adult male body weights ranging from 0.68 to 0.91 kg. The daily food

[47] The FWM performance and input concentrations for TEQs, DDTs, and PCBs are presented and discussed in Appendix E. For aldrin, the model performance was acceptable within the specified criteria described Appendix E using the following calibrated values: spatially weighted average surface sediment concentration = 0.765 µg/kg, average surface water concentration = 1.02 µg/L, and Kow = 6.5.

ingestion rate was estimated as a function of body weight using the following allometric equations developed for carnivorous birds (Nagy 2001):

$$FIR = 3.048 \times BW^{0.665}$$ **Equation 5-4**

Where:

FIR = daily food ingestion rate (g ww/day)
BW = body weight (g)

or,

$$FIR = 0.849 \times BW^{0.663}$$ **Equation 5-5**

Where:

FIR = daily food ingestion rate (g dw/day)
BW = body weight (g)

Using the lower of the average female body weight reported in Dunning (1993; 0.54 kg), the calculated female food ingestion rate was 0.200 kg ww/day and 0.055 kg dw/day and (using Equations 5-4 and 5-5, respectively). The female food ingestion rate calculated based on wet weight was used to estimate dietary exposure doses for hooded merganser.

Incidental Sediment Ingestion Rate

Hooded mergansers are likely to ingest a small amount of sediment incidentally while foraging and indirectly through their prey. An incidental SIR of 2% was used when calculating dietary exposure for hooded mergansers, based on best professional judgment. The female dietary SIR of 0.00110 kg dw/day was calculated assuming 2% incidental sediment ingestion.

Diet Composition

Hooded mergansers feed primarily by diving for whatever small fish are abundant, but they will also eat aquatic invertebrates, especially as hatchlings (Csuti et al. 2001). They are also known to feed on crustaceans, aquatic insects, and small fish (Bendell and McNicol 1995). No data are available on the food preferences of the hooded merganser in Oregon (Marshall et al. 2003). The stomach contents of 138 mergansers from around the United States were presented in Cottam and Uhler (1937), as cited in Dugger et al. (1994). Fish or unidentifiable fish fragments made up 44% of the total stomach contents, and crustaceans (including crayfish) and aquatic insects made up 45%. Mollusks made up less than 1% of the total stomach contents. Prey sizes of fish for hooded merganser have been reported to be 2 inches or less (Alexander 2000).

Hooded mergansers were selected to represent fish-eating water birds in the LWR, birds such as diving water fowl, great blue herons, and other species; therefore, small fish were assumed to make up 70% of the diet. The remaining portion of the merganser diet was represented by clams (25%) and crayfish (5%). These

assumptions are consistent with EPA's October 1, 2004 comments (EPA 2004a) on the dietary assumptions presented in the comprehensive ERA approach technical memorandum (Windward 2004b). Small fish collected in the Study Area included sculpin and juvenile (subyearling) chinook salmon, ranging from 3.6 to 6.8 and 2.0 to 4.7 inches, respectively. In accordance with EPA's comments (dated March 4, 2005) (EPA 2005a) to the dietary assumptions presented in the Ecological PRE approach technical memorandum (Windward 2004a), because juvenile chinook salmon are not resident fish species, peamouth were used as a surrogate species to represent small resident pelagic fish. Therefore, sculpin and peamouth were selected as the fish receptors to represent the fish portion of the merganser diet. Sculpin and peamouth represented 65% and 5% of the total merganser prey diet, respectively. Sculpin are representative resident small fish that are likely an important fish prey item for mergansers. Peamouth were selected to represent resident small pelagic fish and likely represent some portion of the merganser fish diet. The uncertainty of using peamouth (with a body length ranging from 7.1 to 11.1 inches) to represent small fish consumed by merganser is discussed in the uncertainty section (Section 5.4.2).

Site Use Factor

Hooded mergansers are common local breeders in the Willamette Valley with some being year-round residents (Csuti et al. 2001). Therefore, a site use factor of 1.0 was used to estimate the exposure of hooded mergansers in the Study Area. Because mergansers may forage in nearby aquatic and terrestrial environments in addition to the Study Area, a site use factor of 0.5 was also used to estimate risks in the uncertainty section (Section 5.4.2).

Hooded mergansers were assumed to forage in sediment throughout the 9-mile Study Area. Smaller sediment foraging areas (i.e., areas smaller than the Study Area) are possible for mergansers; however, the impact of sediment ingestion on exposure doses for mergansers is very small because incidental sediment ingestion is estimated to be 2%. To evaluate the uncertainty of assuming a Study Area-wide sediment foraging area, a range of sediment EPCs (based on smaller sediment foraging areas) and the impact to hooded merganser exposure doses were calculated and are discussed in the uncertainty section (Section 5.4.2).

Research indicates that mergansers in Wisconsin and Maryland forage to a maximum depth of 0.5 m (Kitchen and Hunt 1969; McGilvrey 1966). Therefore, in order to account for fluctuations in water level and other potential sources of natural variability resulting from tides and river stages, the sediment exposure area for hooded merganser was assumed to be all areas shallower than 1.5 m (approximately 5 ft). To account for the lateral transport of sediments into the shallower habitats, sediments included in the dietary exposure assessment for hooded merganser were limited to surface sediment samples located at depths of 20 ft (NAVD 88) or shallower.

5.2.1.2.3 Bald Eagle

Body Weight and Daily Food Ingestion Rate

Wiemeyer (1991), as cited in EPA (1993), reported average adult female and male body weights for bald eagles to be 4.5 and 3.0 kg, respectively. The food ingestion rate was represented as 12% of the body weight on a wet-weight basis, based on a study by Stalmaster and Gessaman (1982), as presented in EPA (1993), of free-flying eagles in Washington. Using the average female bald eagle body weight, the calculated food ingestion rate was 0.54 kg ww/day.

Incidental Sediment Ingestion Rate

Data on SIRs were not available for bald eagles, but it is likely that bald eagles consume a small amount of sediment while scavenging along the shoreline. An incidental SIR of 2% was used when calculating the dietary exposure for bald eagles based on best professional judgment. The female dietary SIR of 0.00281 kg dw/day was calculated assuming 2% incidental sediment ingestion and 74% moisture in the diet. Seventy-four percent moisture was the average percent moisture in fish prey for bald eagle collected from within the Study Area.

Diet Composition

Bald eagles are opportunistic foragers with site-specific food habits based on available prey species (Buehler 2000; Anthony et al. 1999). In most regions, bald eagles seek out aquatic habitats for foraging and prefer fish (Buehler 2000; Ehrlich et al. 1988). They also eat carrion, various water birds, and small mammals (Csuti et al. 2001). In one study conducted in the lower Columbia River estuary, the diet composition of bald eagles based on direct observation was 90% fish, 7% birds, and 3% mammals (Watson et al. 1991). Other studies conducted in western Washington have reported similar percentages of fish in the bald eagle diet (Knight et al. 1990; Watson and Pierce 1998; Watson et al. 1995). Freshwater fish were the most commonly found prey items in bald eagle nests in the Columbia River Estuary by Watson et al. (1991). The following proportion of fish prey items ingested by bald eagles were reported: 24% Catostomidae (largescale sucker), 20% Clupeidae (American shad), 35.5% Cyprinidae (including common carp and peamouth), 12% Salmonidae (chinook salmon and steelhead), 4% Centrarchidae (including black crappie), and 7% other fish species.

The bald eagle diet (C_{prey}) was estimated using 100% fish species as prey items. Because bald eagles prey primarily on carp and largescale sucker, each of these prey items was selected to represent 45% of the bald eagle diet. Peamouth and northern pikeminnow were each selected to represent 5% of the bald eagle diet because these fish also belong to the Cyprinidae family, a family commonly found in bald eagle nests in the Columbia River Estuary. Chinook salmon tissue were not used to represent salmonids because only juvenile tissue is available from the Study Area, and bald eagles prey on adult salmon. The four selected fish prey species will act as surrogate species, representing a conservative estimate of tissue residues in adult salmon. No tissue data are available for birds and mammals that may serve as prey for

bald eagles. The uncertainty associated with using 100% fish from two primary fish families to represent the bald eagle diet (and excluding birds and mammals) is discussed in the uncertainty section (Section 5.4.2).

Site Use Factor
Bald eagles are known to nest up and down the Willamette River. The closest known nest to the Study Area is on Ross Island at RM 15, though two old nests are located on Sauvie Island (Isaacs and Anthony 2001). Eagles are year-round residents in western Oregon, with some eagles from further north over-wintering in the area (Csuti et al. 2001). Garrett et al. (1993) determined that home ranges in breeding bald eagles in the lower Columbia River Estuary ranged from 5.9 to 47.3 km^2 (with an average home range of 21.7 km^2). The average shoreline range of these home ranges in breeding bald eagles was 5.6 km (3.5 miles). The length of the Portland Harbor Study Area is approximately 14.4 km (9 miles), and it is assumed that breeding bald eagles could potentially consume prey and sediment solely from within the Study Area. Therefore, a site use factor of 1.0 was used to determine dietary exposure for breeding bald eagles. Because non-breeding bald eagles may forage in nearby aquatic and terrestrial environments in addition to the Study Area, a site use factor of 0.5 was also used to estimate risks in the uncertainty section (Section 5.4.2).

Shoreline ranges from the literature indicate that bald eagles may forage in sediment stretches that are smaller than the entire length of the Study Area; however, the impact of sediment ingestion on exposure doses for bald eagles is very small, since incidental sediment ingestion is estimated to be 2%. To evaluate the uncertainty of assuming a Study Area-wide sediment foraging area, a range of sediment EPCs (based on smaller sediment foraging areas) and the impact to bald eagle exposure doses were calculated and discussed in the uncertainty section (Section 5.4.2). Prey tissue for bald eagles from the Study Area cannot be limited to ecologically relevant areas that are smaller than the Study Area because the two species that make up 90% of the bald eagle diet (i.e., carp and largescale sucker) are wide-home ranging fish that forage throughout and outside of the Study Area. Therefore, bald eagles were assumed to forage on sediment and prey throughout the 9-mile Study Area. Sediments included in the dietary exposure assessment for bald eagle were limited to surface sediment samples located at depths of 20 ft (NAVD 88) or shallower because bald eagle sediment exposure is limited to sediment along the shoreline and in very shallow water. The assumption of including sediments from within the 20-ft water depth is evaluated in the uncertainty section (Section 5.4.2).

5.2.1.2.4 Osprey
Body Weight and Daily Food Ingestion Rate
Poole (1983), as cited in EPA (1993), reported average adult female and male osprey body weights during courtship to be 1.88 and 1.48 kg, respectively. The food ingestion rate during the courtship period will be represented as 21% of the body weight on a wet-weight basis, based on studies of adult female osprey in

Massachusetts (Poole 1983) as presented in EPA (1993). Using the average female osprey body weight, the calculated food ingestion rate was 0.395 kg ww/d.

Incidental Sediment Ingestion Rate

Data on osprey sediment ingestion were not available, but osprey may consume a small amount of sediment while scavenging along the shoreline (which occurs very rarely). Therefore, a sediment ingestion of 2% of the dry diet will be assumed for calculating the dietary exposure for osprey. The female dietary sediment ingestion rate of 0.00205 kg dw/day was calculated assuming 2% incidental sediment ingestion and 74% moisture in the diet. Seventy-four percent moisture was the average percent moisture in fish prey for osprey collected from within the Study Area.

Diet Composition

Osprey tend to feed solely on fish, primarily on slow-moving fish that swim near the water surface (Csuti et al. 2001). They may occasionally eat other types of vertebrate prey such as birds, reptiles, and small mammals, and they only rarely feed on invertebrates. Size classes of fish caught by osprey in Idaho were reported by Van Daele and Van Daele (1982): 89% of fish preyed on by osprey were 11 to 30 cm long, suggesting a preference for medium-sized fish by osprey. Henny et al. (2003) investigated the diet of a migratory population of osprey nesting along the main stem of the Willamette River and the McKenzie River in 1993. The following proportions (by biomass) of fish prey items were reported based on the identification of prey items in osprey nests: largescale suckers (83%), northern pikeminnow (7%), common carp (6%), largemouth and smallmouth bass (2%), and invertivorous fish (2%; includes mountain whitefish, black crappie, white crappie, bullhead species, and bluegill). The osprey diet via ingested prey (C_{prey}) was modeled using similar portions of fish prey: 83% largescale sucker, 7% northern pikeminnow, 6% carp, 2% smallmouth bass, 1% black crappie, and 1% brown bullhead.

Site Use Factor

Ospreys are present from March until September, with several breeding pairs nesting in the Portland Harbor Study Area (Henny et al. 2003). In addition to the five pairs nesting between RM 0 and 7.1, osprey also nest in Oaks Bottom Wildlife Refuge at RM 15.4 (Henny et al. 2003). Each fall, they migrate south to western Mexico and Central America (Martell et al. 2001). Because osprey breed in the Study Area, a site use factor of 1.0 was used to estimate exposure, because osprey could be exposed to Study Area contaminants during a critical life stage. Because during the migratory season, non-breeding osprey forage outside the Study Area, a site use factor of 0.5 was also used to estimate risks in the uncertainty section (Section 5.4.2).

Osprey were assumed to forage in sediment throughout the 9-mile Study Area. Smaller sediment foraging areas (i.e., smaller than the Study Area) are possible for osprey; however, the impact of sediment ingestion on exposure doses for osprey is very small, since incidental sediment ingestion is estimated to be 2%. To evaluate the uncertainty of assuming a Study Area-wide sediment foraging area, a range of

sediment EPCs (based on smaller sediment foraging areas) and the impact to osprey exposure doses were calculated and discussed in the uncertainty section (Section 5.4.2). Prey tissue for osprey from the Study Area cannot be limited to ecologically relevant areas that are smaller than the Study Area because largescale suckers, the primary prey species for osprey, are wide-home ranging fish that forage throughout and outside of the Study Area. Therefore, osprey were assumed to forage on sediment and prey throughout the 9-mile Study Area. Sediments included in the dietary exposure assessment for osprey were limited to surface sediment samples located at depths of 20-ft (NAVD 88) or shallower because osprey sediment exposure is limited to sediment along the shoreline and in very shallow water. The assumption of including sediments from within the 20 ft water depth is evaluated in the uncertainty section (Section 5.4.2).

5.2.1.2.5 Mink

Body Weight and Daily Food Ingestion Rate

Hornshaw et al. (1983), as presented in EPA (1993), reported average farm-raised adult female and male BWs for mink in the summer to be 0.974 and 1.734 kg, respectively. The daily food ingestion rate was estimated as 16% and 12% of body weight on a wet weight basis, based on studies of farm-raised female and male mink in Michigan (Bleavins and Aulerich 1981), as presented in EPA (1993). Using the female mink parameters, the calculated food ingestion rate for females was 0.156 kg ww/d.

Incidental Sediment Ingestion Rate

Data were not available on the amount of sediment consumed by mink while feeding. Beyer et al. (1994), as presented in EPA (1993), reported a soil ingestion of 9.4% of the dry diet for raccoons. The female dietary sediment ingestion rate of 0.00381 kg dw/day was calculated assuming 9.4% incidental sediment ingestion and 74% moisture in the diet. Seventy-four percent moisture was the average percent moisture in crayfish and fish prey for mink collected from within the Study Area. This conservative incidental sediment ingestion rate assumed for mink is evaluated in the uncertainty section (Section 5.4.2).

Diet Composition

Mink are opportunistic feeders and consume a range of prey including muskrats, fish, frogs, crayfish, small mammals, and birds found near water (Csuti et al. 2001). The prey items of mink are largely dependent on availability (Melquist et al. 1981; Racey and Euler 1983; Ward et al. 1986; Wise et al. 1981), and portions of fish in the mink diet vary widely across field studies. In Idaho, fish made up 59% of the mink diet, primarily cyprinids (Melquist et al. 1981); and in Michigan, fish made up 85% of the mink diet (ww), based on stomach contents. In other field studies, fish remains made up a smaller portion of the mink diet, with higher portions of birds, crayfish, and mammals reported (Ward et al. 1986; Wise et al. 1981). Region-specific studies on the quantitative portions of mink prey items were not available. The Washington Department of Game (WDG; now called Washington Department of Fish and

Wildlife) reported that crayfish, fish, birds, and mammals are important foods of mink in the Columbia River study area (WDG 1980). The most frequently eaten fish throughout the year that were identified in the Columbia River study were carp, sculpin, suckers, and centrarchids (WDG 1980); however, many fish consumed in this study were not identified. Other field studies have reported that mink do not appear to select fish prey from a particular size range (Ward et al. 1986; Wise et al. 1981).

Based on available prey tissue data collected in Round 1 sampling, the mink diet was modeled using prey items from crayfish and fish species collected from the Study Area. Because of the opportunistic nature of mink and based on the available data, all fish species collected in Round 1 were used to model the mink diet. The following prey items were assumed to be the primary prey items of the mink (these species were commonly observed in regional studies) and each made up 15% diet via ingested prey (C_{prey}): crayfish, largescale sucker, carp, sculpin, and smallmouth bass. The remaining fish, peamouth, juvenile chinook salmon, northern pikeminnow, brown bullhead, and black crappie, each made up 5% of the mink diet.

Site Use Factor

Mink are semi-aquatic mammals that are closely associated with water and riparian habitats. They are active year-round and are primarily nocturnal. A site use factor of 1.0 was assumed for mink. Mink have linear home ranges along rivers and move considerable distances in streams and on river banks (Csuti et al. 2001). Environment Canada (2003) reported mink home ranges as generally restricted to areas adjacent to bodies of water, with American mink males ranging over 2.5 to 5.5 km (1.6 to 3.4 miles) of shoreline, and female mink ranging over a more restricted 0.5 to 3 km (0.3 to 1.7 miles) of shoreline.

For the purposes of this risk evaluation, mink were assumed to forage on sediment and prey throughout the 9-mile Study Area. The literature indicate that mink may forage in smaller sediment stretches than the entire length of the Study Area; however, the impact of sediment ingestion on exposure doses for mink is expected to be small, since incidental sediment ingestion is estimated to be 9%. To evaluate the uncertainty of assuming a Study Area-wide sediment foraging area, a range of sediment EPCs (based on smaller sediment foraging areas) and the impact to mink exposure doses were calculated and discussed in the uncertainty section (Section 5.4.2). Prey tissue for mink from the Study Area cannot be limited to ecologically relevant areas that are smaller than the Study Area because some prey species for mink include wide-home ranging fish that forage throughout and outside of the Study Area. Sediments included in the dietary exposure assessment for mink were limited to surface sediment samples located at depths of 20-ft (NAVD 88) or shallower because mink sediment exposure is limited to sediment along the shoreline and in very shallow water.

5.2.1.2.6 River Otter

Body Weight and Daily Food Ingestion Rate

Average adult female and male river otter body weights in western Oregon and Washington have been reported for trapped otters submitted to the US Geological Service (Grove 2004). Body weights were reported without pelts, and weights were adjusted to estimate body weight with pelts using the following equation, as agreed upon by EPA, EPA's partners, and LWG in the preparation of the Ecological PRE (Windward 2005):

$$BW_{unpelted} = BW_{pelted} + \frac{0.5748}{0.8437}$$ **Equation 5-6**

Where:

$BW_{unpelted}$ = body weight of river otter with skin (kg)
BW_{pelted} = body weight of river otter without skin (kg)

Estimated pelted body weights for adult female and male river otters were 9.46 and 11.15 kg, respectively. Estimated pelted body weights for juvenile female and male river otters were 7.7 and 8.48 kg, respectively. Juvenile female body weight was used to estimate exposure to river otter. This body weight represent a more conservative value and are similar to adult body weight in other regions; Melquist and Hornnocker (1983), as presented in EPA (1993), reported average female and male river otter BWs in Idaho as 7.9 and 9.2 kg, respectively. The daily food ingestion rate for river otter was estimated as a function body weight using the following allometric equation developed for carnivorous mammals (Nagy 2001):

$$FIR = 0.102 \times BW^{0.864}$$ **Equation 5-7**

Where:

FIR = daily food ingestion rate (g dw/day)
BW = body weight (g)

or,

$$FIR = 0.348 \times BW^{0.859}$$ **Equation 5-8**

Where:

FIR = daily food ingestion rate (g ww/day)
BW = body weight (g)

Using the female juvenile river otter body weight, the calculated female food ingestion rate was 0.233 kg dw/day and 0.759 kg ww/day. The female food ingestion rate calculated based on wet weight was used to estimate dietary exposure doses for river otter.

Incidental Sediment Ingestion Rate
No data are available concerning sediment ingestion for the river otter. River otters may ingest a small amount of sediment incidentally while foraging and indirectly through their prey. Therefore, an estimated SIR of 2% of the dry diet will be used when calculating the dietary exposure dose for river otter. The female dietary SIR of 0.00465 kg dw/day was calculated assuming 2% incidental sediment ingestion.

Diet Composition
River otters are opportunistic carnivores that take advantage of food that is most abundant and easiest to catch, although fish are their primary prey (EPA 1993). Slower-moving fish such as suckers, carp, chubs, and bullheads are generally eaten most frequently (Wise et al. 1981; Kurta 1995). Other components of the river otter's diet may include aquatic invertebrates (including crayfish, mussels, clams, and aquatic insects), frogs, snakes, turtles, and occasionally scavenged small mammals and birds (Coulter et al. 1984; Csuti et al. 2001). Field studies in British Columbia, western Oregon, and Alaska identifying the prey contents in river otter scat or stomach contents have found that fish remains make up the majority of the prey items, occurring more than 85% of the time (Stenson et al. 1984; Larsen 1984; Toweill 1974). River otters specialize in fish predation, whereas mink are more generalized carnivores, preying on fish, birds, and mammals (Wise et al. 1981). This is supported by the results of Toweill (1974) in which prey items were identified in river otter stomachs. Crustaceans (most commonly crayfish) were observed in 20% of otter stomachs analyzed, with amphibians and birds occurring less frequently. Mollusks were observed in only 2% of the stomachs examined. The proportion of prey types consumed by river otter will be based on that reported by Larsen (1984) for southeastern Alaskan river otter: 86% fish, 10% crab, 2% invertebrates other than crab, 1% birds, and 1% mammals and plant material. This study will be used because it was the only study from the Pacific Northwest that reported remains in scat on a volume basis rather than as a frequency of occurrence or as a qualitative observation. Larsen (1984) reported the following proportions of prey ingested by river otter: 86% fish, 10% crab, 2% invertebrates other than crab, 1% birds, and 1% mammals and plant material. Thus, for estimating prey concentration (C_{prey}), river otters were assumed to consume 88% fish, 10% crayfish, and 2% clams.

Fish in the river otter diet will be represented by those fish commonly observed in region-specific field studies. The primary prey items of river otter in the Columbia River in the summer were carp, crayfish, suckers and centrarchids (sunfishes, including bass and perch). Other prey (including sculpin, American shad, northern pikeminnow, salmon, birds, mammals, insects, and mollusks) were eaten less frequently and were considered of minor importance (WDG 1984). Henny et al. (1996) and the Columbia Basin Fish and Wildlife Authority (CBFWA) (1996) reported carp and crayfish as the most frequently eaten prey of river otter in the summer in areas within the Lower Columbia River. Another study reported that carp and sculpin were most frequently preyed upon by river otter in the Columbia River estuary in the spring (WDG 1984). Based on these region-specific studies, the fish

tissue used to model the river otter diet included carp, sculpin, largescale sucker, and centrarchids (smallmouth bass and black crappie). Carp, sculpin, and largescale sucker each made up 26% of the diet, and smallmouth bass and black crappie will each make up 5% of the diet.

Site Use Factor

During the breeding season, river otters may travel considerable distances over land (Csuti et al. 2001). A study of river otters in Idaho found the home range length along streams for juvenile, yearling, and adult river otter averaged 39.8 km (24.7 miles) (Melquist and Hornocker 1983). River otters range over an area sufficiently large enough for foraging and reproduction (Melquist and Dronkert 1987); however, they are typically found in a limited number of activity centers within their overall range. At any given time, river otters generally occupy only a few kilometers of stream but often move from one area to another (Nebraska Game and Parks Commission 2000). A radio-tracking study of relocated river otters was conducted as part of the New York State Department of Environmental Conservation river otter reintroduction program. This study showed that river otter ranges were between 1.5 and 22.4 km long, with an average distance of 10 km (6 miles) for individuals monitored in western New York State (Spinola et al. 1999, as cited in EPA 1993). Because of the average 10 to 40-km linear length documented in the literature, and because the extent of the Study Area is approximately 9 miles (14.5 km), it was assumed that river otters could potentially consume prey and sediment solely from within the Study Area. Therefore, a site use factor of 1.0 was assumed for river otters.

The literature indicate that river otter may forage in smaller sediment stretches than the entire length of the Study Area; however, the impact of sediment ingestion on exposure doses for river otter is very small, since incidental sediment ingestion is estimated to be 2%. To evaluate the uncertainty of assuming a Study Area-wide sediment foraging area, a range of sediment EPCs (based on smaller sediment foraging areas) and the impact to river otter exposure doses were calculated and discussed in the uncertainty section (Section 5.4.2). Prey tissue for river otter from the Study Area cannot be limited to ecologically relevant areas that are smaller than the Study Area because the various prey species for river otter include wide-home ranging fish that forage throughout and outside of the Study Area. Sediments included in the dietary exposure assessment for river otter were limited to surface sediment samples located at depths of 20 ft (NAVD 88) or shallower because river otter sediment exposure is limited to sediment along the shoreline and in very shallow water.

5.2.1.3 Prey Tissue and Surface Sediment EPCs

This section presents the Round 2 COPC concentrations in prey tissue and surface sediment that were used in Equations 5-1 and 5-3 to estimate dietary doses for each of the wildlife receptors. Exposure dose estimates for each wildlife Round 2 COPC/receptor pair were based on an estimate of each Round 2 COPC EPCs in each of the various media (i.e., surface sediment and prey tissue). EPCs were calculated for

all Round 2 COPCs in surface sediment and for each of the prey tissue items as an upper bound estimate (e.g., a 95 UCL).

ProUCL software was developed by EPA to compute an appropriate UCL of an unknown population mean. ProUCL tests for normality, lognormality, and a gamma distribution of the dataset and computes a conservative and stable UCL of the unknown population mean (EPA 2004c). Prior to running data through ProUCL to determine UCLs, non-detected data were adjusted to one half the detection limit to represent the concentration value. For each Round 2 COPC, the UCL recommended by ProUCL was used at the EPC for the risk calculations. In the case where the insufficient data were available, the maximum concentration[48] was used to represent the EPC. Attachment G1 presents the summary statistics (i.e., minimum, maximum, and mean Round 2 COPC concentrations), distribution types, ProUCL recommended UCLs, and selected EPCs for each Round 2 COPC.

5.2.1.3.1 Prey Tissue

Data were available for 12 tissue types that are potential prey of wildlife receptors in the Study Area: clams, crayfish, sculpin, peamouth, carp, largescale sucker, northern pikeminnow, smallmouth bass, black crappie, brown bullhead, juvenile chinook salmon, and worms (from sediment bioaccumulation tests). These data are described in Section 2.0.

For all wildlife receptors, except for spotted sandpiper, if was assumed that the foraging area includes the entire Study Area based on the home range information presented above. Thus, for each prey type ingested by those species, the EPC was calculated using all data from the Study Area as one exposure dataset. The EPC for each tissue type was estimated as the recommended ProUCL value or maximum concentration, when insufficient data were available. The EPCs for all Round 2 COPCs in each tissue type used to estimate dietary exposure doses for hooded merganser, bald eagle, osprey, mink, and river otter, are presented in Table 5-10.

For spotted sandpiper, it was assumed that each individual beach area represented a foraging area. At each beach area, a clam and worm tissue EPC was represented as the maximum tissue concentration collected within that beach habitat area. UCLs were not calculated because the sample size ($n \leq 2$) was too small at all beach locations except at B17 where four worm tissue samples were available. Where tissue data were unavailable, clam and worm tissue concentrations were estimated using the FWM or a site-specific regression, or by a UCL, as described in Section 5.2.2.2. In addition to the beach by beach analysis, a Study Area-wide scenario was evaluated. In this scenario, the clam and worm tissue EPC was estimated as the recommended ProUCL value. The EPCs for all Round 2 COPCs in worm and clam tissue at each beach area and Study Area-wide for spotted sandpiper are presented in Tables 5-11 through 5-13.

[48] Where the maximum concentration was a non-detected value, the full detect limit was used to represent the EPC.

5.2.1.3.2 Surface Sediment

Surface sediment data were used to estimate Round 2 COPC exposure results from incidental sediment ingestion. For hooded merganser, osprey, bald eagle, mink, and river otter, it was assumed that the foraging areas includes the entire Study Area within the 20-ft depth (NAVD 88) or shallower; thus, EPC for surface sediment were calculated as the lower of the UCL or maximum concentration of all surface sediment data within this water depth. The sediment EPCs used to estimate dietary exposure doses for hooded merganser, bald eagle, osprey, mink, and river otter, are presented in Table 5-14.

For spotted sandpiper, if was assumed that each individual beach area represented a foraging area. At each beach area, a sediment EPC was represented as the maximum tissue concentration collected within that beach habitat area.[49] In addition to the beach by beach analysis, an all beach scenario was evaluated. In this scenario, the sediment EPC was estimated as the recommended ProUCL value. The sediment EPCs for all Round 2 COPCs at each beach area and all beach for spotted sandpiper are presented in Tables 5-15 and 5-16.

5.2.1.4 Estimated Dietary Doses

Exposure dietary dose estimates were calculated for each wildlife Round 2 COPC/receptor pair based on the information presented in preceding sections. Estimated dietary dose exposures for the other wildlife receptors (i.e., hooded mergansers, bald eagle, osprey, mink, and river otter) are presented in Table 5-17. Estimated dietary dose exposures for spotted sandpiper are presented in Table 5-18 through 5-20.

5.2.2 Round 2 COPC Effects Assessment

This section presents a summary of the toxicity literature for each of the Round 2 COPCs and the selected TRVs for bird and mammal Round 2 COPCs. NOAEL and LOAEL TRVs were presented in the TRV Technical Memorandum (Appendix B of the Ecological PRE; Windward 2005) based on an extensive search on the available toxicological literature for the development of bird and mammalian wildlife dietary TRVs and were selected to represent conservative screening level toxicological thresholds. EPA's comments to the Ecological PRE state that "overall, [avian and mammalian dietary] TRVs selected were the most conservative of those acceptable studies and...are reflective of EPA's comments for deriving adequately conservative dietary TRVs" (EPA 2006b). EPA provided further direction (EPA 2006a, e) on specific wildlife dietary dose TRVs that should be changed for this Round 2 ERA, and these changes were incorporated to the NOAEL TRVs used for identifying Round 2 COPCs in the risk screening step (Section 5.2.1). However, for the purposes of risk characterization (Section 5.3), EPA's recommendation of the use of ecological soil screening levels (Eco SSLs) to evaluate exposure of wildlife receptors to metals

[49] UCLs were not calculated because the sediment sample size at all beach locations was too small ($n \leq 2$ at all beach locations except at B24 where four sediment samples were available).

were not considered appropriate by LWG. Eco SSLs are based on multiple exposure pathways (i.e., dietary, gavage, drinking water). Non-dietary exposure pathways such as gavage and drinking water are not preferred over dietary studies for developing toxicity thresholds for evaluating dietary risks to wildlife receptors, when such literature is available. Drinking water is considered a minor pathway for wildlife receptors and the bioavailability of chemicals from water may be different from that of food, and therefore, these studies were not considered representative of exposure in the Study Area. Therefore, Eco SSLs were not used as TRVs for risk characterization; dietary TRVs based on the toxicological literature were used for risk characterization of Round 2 COPCs.

Changes to wildlife dietary TRVs based on EPA comments (EPA 2006a, b, e), other than the use of Eco SSLs, were adopted for risk characterization of Round 2 COPCs in this Round 2 ERA and incorporated into the selection of NOAEL and LOAEL TRVs. The details on assumptions of body weights and food ingestion rates used to calculate dose TRVs from the literature are also presented in the TRV Technical Memorandum (Appendix B of the Ecological PRE; Windward 2005). All TRVs recommended in the TRV Technical Memorandum were used to characterize wildlife exposure, with the exception of those presented in the TRV Technical Memorandum Addendum (Attachment G5). Changes to selected TRVs for the Round 2 ERA were based on the following:

- Changes to assumptions made in dietary dose TRVs changed the value of selected TRV or resulted in lower TRVs than those selected in the TRV Technical Memorandum
- Additional review of toxicological studies since the preparation of the TRV Technical Memorandum

Toxicological data presented in this section are assessed in combination with exposure data (presented in Section 5.2.1) in the risk characterization (Section 5.3). Region 10 UFs presented in Attachment G6 were used to extrapolate NOAEL TRVs where no toxicological data were available lower than the selected LOAEL TRVs.

It should be noted that the selected NOAEL and LOAEL TRVs are conservatively based, and some may not be appropriate for use in the BERA. The uncertainty of selected wildlife TRVs are presented in each of the subsections and recommendations for BERA TRVs are also presented, where appropriate.

5.2.2.1 Bird TRVs

This section presents a brief summary of the laboratory and field toxicity studies reviewed for Round 2 COPCs identified for bird receptors (Table 5-5), and selected TRVs used to estimate risks. Details on the TRV selection process and on the studies reviewed are presented in the TRV Technical Memorandum and in the TRV Technical Memorandum Addendum (Attachment G5). TRVs for PCBs were selected

for both total PCBs (generally based on Aroclors) and for 2,3,7,8,-TCDD (as TEQs for dioxins and furans [dioxin TEQ], and dioxin-like PCB congeners [PCB TEQ]).

A single NOAEL and LOAEL TRV were selected for each Round 2 COPC and used for all bird receptors.

5.2.2.1.1 Dioxin TEQ and PCB TEQ

The dioxin TEQ and PCB TEQ exposure doses for each bird receptor species was compared to the TRVs selected for 2,3,7,8-TCDD. Two studies in which birds were exposed to 2,3,7,8-TCDD via daily oral intubation and weekly intraperitoneal injections were reviewed (Table 5-21). The LOAEL and NOAEL of 0.14 and 0.014 µg/kg bw/day) (Nosek et al. 1992), respectively, were selected as the dietary TRVs for 2,3,7,8-TCDD in birds because they represent the lower-effect-level concentration of the two studies considered. In addition, the selected TRVs are based on a study measuring adverse effect over a chronic exposure (10 weeks) during a critical life stage; the other study reviewed was not conducted during a critical life stage or over a chronic period.

There is high uncertainty associated with the selected TRVs because they are based on injection exposure. Intraperitoneal injection is not directly relevant to the dietary exposure pathway for birds, and the selected TRVs may not accurately predict dietary dioxin toxicity.

5.2.2.1.2 Total PCBs

Twenty-three studies that considered the oral toxicity of PCB Aroclors to birds via food or capsule ingestion were evaluated for TRV selection (Table 5-22).The lowest LOAEL calculated for chickens of 0.58 mg/kg bw/day from Britton and Huston (1973) was selected as the LOAEL TRV. The lowest LOAEL of 0.35 mg/kg bw/day, calculated from Lowe and Stendell (1991), was not selected because the impact of eggshell thinning on kestrel reproductive success was not reported, whereas reproductive success of chickens measured in Britton and Huston (1973) is clearly impaired because significant effects on hatchability were observed in the treated group compared to the control group. It should be noted that there is not a large difference between the calculated LOAELs reported in these two studies (i.e., 0.58 vs. 0.35 mg/kg bw/day). The NOAEL of 0.29 mg/kg bw/day, calculated from the same study (Britton and Huston 1973) as the selected LOAEL, was selected as the NOAEL TRV. Hatchability was not affected in chickens at this dietary concentration or at the lower concentrations reported in Ahmed et al. (1978) and Scott et al. (1975).

Chickens appear to have a higher sensitivity to PCBs than do other avian species in the laboratory and may not be representative of Portland Harbor avian receptors. Thus, there is some uncertainty associated with the selected TRVs for PCBs in birds. However, the selected TRVs represent conservative values reported in the available literature. For the BERA, TRVs based on non-chicken (and non-domestic) species may be more appropriate.

5.2.2.1.3 Arsenic

EPA and its partners recommended the use of Eco SSLs for those metals where SSLs are available for birds(EPA 2006a). The bird Eco SSL for arsenic, 2.24 mg/kg bw/day, was used for screening and Round 2 COPC identification (Section 5.2.1). For the purposes of risk characterization, Eco SSLs were not considered appropriate (see above for rationale), and dietary studies from the original toxicological literature were considered more representative, and therefore, more appropriate.

Five studies on the dietary toxicity of arsenic to birds were evaluated for TRV selection (Table 5-23). The lowest LOAEL and NOAEL of 6.8 and 2.3 mg/kg bw/day, respectively, based on cowbird mortality were selected as the TRVs for risk evaluation because they represent the most conservative values calculated from the literature reviewed (USFWS 1969) and represent a dietary exposure route.

However, there is high uncertainty associated with the selected dietary TRVs because in the selected study, cowbirds were exposed to a chemical mixture of both arsenic and copper. There is additional uncertainty associated with the selected TRVs because the original source could not be reviewed. The next highest LOAEL (40 mg/kg bw/day) and associated NOAEL (10 mg/kg bw/day) based on Stanley et al. (1994) likely represent more appropriate toxicological thresholds for arsenic exposure to birds because birds in the study were exposed only to arsenic. These TRVs are recommended as more appropriate TRVs for the BERA.

5.2.2.1.4 Cadmium

EPA and its partners recommended the use of EPA Eco SSLs for those metals where data were available for birds (EPA 2006a). The bird Eco SSL for cadmium, 1.47 mg/kg bw/day, was used for screening and Round 2 COPC identification (Section 5.2.1). For the purposes of risk characterization, Eco SSLs were not considered appropriate (see above for rationale), and dietary studies from the original toxicological literature were considered more representative, and therefore, more appropriate.

Nine laboratory studies on the toxicity of dietary cadmium to birds were evaluated for TRV selection (Table 5-24). The LOAEL and NOAEL of 2.9 and 0.73 mg/kg bw/day, respectively, were selected as the TRVs for cadmium. The selected LOAEL of 2.9 mg/kg bw/day was based on the study resulting in the lowest LOAEL where a clear dose-response relationship was observed (Leach et al. 1979) and where birds were exposed via the dietary pathway only. The selected NOAEL of 0.73 mg/kg bw/day was calculated from this same study.

There is high uncertainty associated with the selected TRVs for cadmium. The selected TRVs are based on eggshell thinning in chickens. Egg production in a domestic species is not an appropriate endpoint for TRV selection (see the TRV Technical Memorandum) and it is unknown if the degree to which eggshell thinning observed in chickens would affect other non-domestic species and reproductive success. The next highest LOAEL with a dose-response effect (4.0 mg/kg bw/day)

based on Richardson et al. (1974) likely represents a more appropriate toxicological threshold for cadmium exposure to evaluate risks to bird receptors in the LWG Study Area. At this dose, growth was affected in Japanese quail chicks. The lowest growth NOAEL of 1.6 mg/kg bw/day (Cain et al. 1983) below this LOAEL also likely represents a more appropriate no-effects toxicological threshold. These NOAEL and LOAEL TRVs of 1.6 and 4.0 mg/kg bw/day, respectively, are recommended as more appropriate TRVs for the BERA.

5.2.2.1.5 Chromium

EPA and its partners recommended the use of EPA Eco SSLs for those metals where SSLs are available for birds (EPA 2006a). The bird Eco SSL for chromium, 2.66 mg/kg bw/day, was used for screening and Round 2 COPC identification (Section 5.2.1). For the purposes of risk characterization, Eco SSLs were not considered appropriate (see above for rationale), and dietary studies from the original toxicological literature were considered more representative, and therefore, more appropriate.

Five studies that exposed birds to dietary chromium were evaluated for TRV selection and one study that exposed birds to chromium via intraperitoneal injection was reviewed (Table 5-25). The LOAEL and NOAEL of 5.0 and 1.0 mg/kg bw/day, respectively, were selected as the TRVs for chromium. The selected LOAEL TRV was the lowest dietary LOAEL calculated from the literature reviewed, and the selected NOAEL TRV was the lowest acceptable dietary NOAEL calculated.

There is uncertainty associated with the TRV literature dataset for chromium because of the limited number of dietary studies reporting adverse effects and the high variability of these effect levels. In addition, there is uncertainty associated with the selected TRVs because the original source could not be reviewed.

5.2.2.1.6 Copper

EPA and its partners recommended the use of EPA Eco SSLs for those metals where SSLs are available for birds (EPA 2006a). The bird Eco SSL for copper, 4.05 mg/kg bw/day, was used for screening and Round 2 COPC identification (Section 5.2.1). For the purposes of risk characterization, Eco SSLs were not considered appropriate (see above for rationale), and dietary studies from the original toxicological literature were considered more representative, and therefore, more appropriate.

Seven studies on the toxicity of dietary copper to birds were evaluated for TRV selection (Table 5-26). Only one of the studies evaluated the chronic effects (i.e., over 10 weeks) of dietary copper exposure and the LOAEL and NOAEL of 62 and 47 mg/kg bw/day, respectively, were selected from this study (Mehring et al. 1960). The TRVs were selected from this study because TRVs based on chronic exposures are preferred when data are available (see the TRV Technical Memorandum). The highest NOAEL and LOAEL were calculated from Mehring et al. (1960); however, growth was affected in the other two studies during a subchronic period (4 weeks), and it is unknown whether the adverse effects on growth would persist over time.

5.2.2.1.7 Lead

EPA and its partners recommended the use of EPA Eco SSLs for those metals where SSLs are available for birds (EPA 2006a). The bird Eco SSL for lead, 1.63 mg/kg bw/day, was used for screening and Round 2 COPC identification (Section 5.2.1). For the purposes of risk characterization, Eco SSLs were not considered appropriate (see above for rationale), and dietary studies from the original toxicological literature were considered more representative, and therefore, more appropriate.

A total of 14 studies on the potential adverse effects of lead on birds were reviewed. (Table 5-27). The LOAEL and NOAEL of 20 and 2.0 mg/kg bw/day, respectively, were selected as the TRVs. The selected LOAEL was the lowest dietary LOAEL based on an appropriate endpoint (lower LOAELs were reported for egg production in domestic quail). The selected LOAEL represents a conservative level based on quail. Domestic species, such as Japanese quail, may have a higher sensitivity to lead, and the selected LOAEL may overestimate risk to the selected avian receptors in Portland Harbor. It appears from the reviewed studies that common terns, herring gulls, and American kestrels are less sensitive to lead, even during chick development. The selected NOAEL was reported in the same study and represents the lowest acceptable dietary NOAEL reported in the literature reviewed.

5.2.2.1.8 Mercury

Fourteen studies measuring the toxicity of dietary mercury via injection or oral ingestion were reviewed for TRV selection (Table 5-28). The LOAEL of 0.050 mg/kg bw/day mercury was selected as the LOAEL. No dietary dose that was lower than this selected LOAEL was calculated from the literature, so a dietary NOAEL TRV was extrapolated from the selected LOAEL using an UF of 5. The resulting NOAEL TRV was 0.010 mg/kg bw/day.

The selected LOAEL represented the most conservative number available from the literature reviewed; however, it is uncertain because the literature did not always report adverse effects at this level. Therefore, the selected LOAEL may or may not always result in an adverse effect. There is also uncertainty with the use of a UF to extrapolate a chronic NOAEL TRV from a chronic LOAEL TRV.

5.2.2.1.9 Selenium

Five studies that exposed either screech owls or mallards to selenium through diet were reviewed and evaluated for TRV selection (Table 5-29). The LOAEL and NOAEL of 0.82 and 0.42 mg/kg bw/day, respectively, were selected as the LOAEL and NOAEL for dietary selenium in birds. The TRVs are based on the bird species with the highest sensitivity (mallards).

5.2.2.1.10 Thallium

Only two studies were reviewed on thallium toxicity to birds (Table 5-30). The selected LOAEL TRV of 24 mg/kg bw/day was based on Hudson et al. (1984). This was the lowest LOAEL reported in the reviewed literature; however, there is high uncertainty associated with this selected LOAEL because high mortality was

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observed over a short exposure period (14 days). No NOAEL was reported in the literature reviewed below the selected LOAEL, so a NOAEL TRV was extrapolated from the selected LOAEL TRV using a UF of 50. The resulting NOAEL TRV was 0.48 mg/kg bw/day. There is high uncertainty associated with the literature dataset for thallium toxicity to birds due to the paucity of studies available.

5.2.2.1.11 Zinc

Six studies on zinc toxicity to birds were evaluated for TRV selection (Table 5-31). The lowest dietary LOAEL, 124 mg/kg bw/day, was selected as the LOAEL TRV, where the growth of chicks was affected following dietary exposure to zinc over 5 weeks. In the same study, a NOAEL of 82 mg/kg bw/day was reported and selected as the NOAEL TRV.

There is uncertainty associated with the selected LOAEL because the growth effects observed in chicks did not take place over a chronic exposure period or during a critical life stage, and it is unknown whether or not the adverse effect on growth would persist over time. However, none of the studies reporting an effect level concentration was conducted over a chronic exposure or during a critical life stage. There is additional uncertainty with the use of a UF to extrapolate a chronic NOAEL TRV from a subchronic NOAEL TRV.

5.2.2.1.12 Tributyltin

Two studies on TBT toxicity to birds were evaluated for TRV selection (Table 5-32).The LOAEL and NOAEL of 3.6 and 1.4 mg/kg bw/day, respectively, were selected as the TRVs.

There is some uncertainty associated with the selected TRVs because the literature dataset for TBT toxicity to birds is limited to two studies reporting the effects in Japanese quail only. The sensitivity of other bird species to TBT is unknown.

5.2.2.1.13 PAHs: Benzo(a)pyrene and Total PAHs

Two studies on the dietary toxicity of benzo(a)pyrene (Table 5-33) and one study on the dietary toxicity of PAH mixtures (Table 5-34) to adult birds were reviewed. The LOAEL of 1.4 mg/kg bw/day benzo(a)pyrene was selected as the LOAEL TRV. This was the lowest LOAEL and was selected as the LOAEL TRV for benzo(a)pyrene in birds. A NOAEL TRV for benzo(a)pyrene (0.28 mg/kg bw/day) was extrapolated from the selected LOAEL TRV using a UF of 5. For total PAHs, no LOAELs were reported in the reviewed studies. The lowest NOAEL of 40 mg/kg bw/day was selected as the NOAEL TRV. At 40 mg/kg bw/day, growth was adversely affected in mallards fed an petroleum hydrocarbon mixture containing PAHs[50] combined with paraffin wax at three months, however, at seven months, growth was recovered and change in body weight was not significantly different than the control group (Patton

[50] Aromatic hydrocarbon mixture contained the following: ethylbenzene, 1,2,3,4-tetrahydronaphthalene, dimethylnaphthalene, 2,3,3-trimethylindolenine, acenaphthylene, acenaphthlene, phenanthrene, 2-methylbenzothiazole, dibenzothiophene, and 2,6-dimethylquinoline.

and Dieter 1980). The reduction observed in growth at 3 months was attributed to food avoidance of the aromatic hydrocarbon mixture feed because of the noxious odor of the petroleum hydrocarbons.

There is high uncertainty associated with the selected PAH TRVs for birds. For benzo(a)pyrene, there is uncertainty associated with the selected LOAEL due to the route of exposure (intramuscular injection). Injection is not directly relevant to the dietary exposure pathway for birds, and the selected TRVs may not accurately predict dietary PAH toxicity to birds. In addition, there is uncertainty associated with the frequency of exposure (weekly) and the impact of the measured endpoints (fertility and ovarian appearance) on actual reproductive success at the selected LOAEL. It is unknown whether the effects on fertility would reduce the reproductive potential of pigeons at the population level. For total PAHs, the selected NOAEL TRV is based on exposure to a petroleum hydrocarbon mixture that contained some individual PAHs. Therefore, birds were exposed to a PAH mixture, in addition to other chemicals. In addition, food avoidance was observed in the mixture PAH study and no LOAEL TRV was derived.

Finally, there is uncertainty with the literature dataset for both the benzo(a)pyrene and PAH mixtures due to the limited number of studies, the high variability of the effect levels reported, and the use of a UF to extrapolated a chronic benzo(a)pyrene NOAEL TRV from a chronic LOAEL TRV.

5.2.2.1.14 Aldrin

Two studies that measured the toxicity of aldrin to birds were reviewed and evaluated for TRV selection (Table 5-35). The lowest LOAEL calculated from the reviewed literature of 0.04 mg/kg bw/day was selected as the TRV for aldrin. A NOAEL TRV for aldrin was extrapolated from the selected LOAEL TRV using a UF of 5. The resulting NOAEL TRV, 0.008 mg/kg bw/day, was selected for aldrin.

There is some uncertainty associated with the selected LOAEL TRV because the literature dataset for aldrin toxicity in birds is limited to two studies reporting the effects in domestic species only, with high variability in the dose levels resulting in adverse effects. The sensitivity of other bird species to aldrin is unknown. There is additional uncertainty with the use of a UF to extrapolate a chronic NOAEL TRV from a subchronic NOAEL TRV.

5.2.2.1.15 DDTs: Sum DDD, Sum DDE, Sum DDT, Total DDTs

For oral toxicity to DDT, sufficient data were available from the literature to derive separate TRVs for sum DDD, sum DDE, sum DDT, and total DDT (or a mixture of DDT and its metabolites). These toxicological data are presented in the following subsections.

DDD

Two studies on the oral toxicity of DDD to birds were evaluated for TRV selection (Table 5-36). The lowest LOAEL of 0.90 mg/kg bw/day was selected as the LOAEL

for DDD. A NOAEL TRV for DDD was extrapolated from the selected LOAEL TRV using a UF of 5. The resulting NOAEL TRV, 0.18 mg/kg bw/day, was selected for DDD.

There is uncertainty associated with the selected LOAEL TRV because the literature dataset was limited to two studies. There is additional uncertainty with the use of a UF to extrapolate a chronic NOAEL TRV from a chronic LOAEL TRV.

DDE
Seventeen studies evaluating dietary DDE toxicity to birds were evaluated for TRV selection (Table 5-37) the lowest LOAEL of 0.32 mg/kg bw/day (calculated from Mendenhall et al. 1983) was selected as the LOAEL TRV. The selected LOAEL is a conservative threshold effect level based on the most sensitive species tested in the reviewed literature. Based on EPA's comments to the Ecological PRE (EPA 2006e), A selected NOAEL TRV for DDE (0.064 mg/kg bw/day) was extrapolated from the selected LOAEL TRV using a UF of 5.

LWG recommends the use of actual toxicity data and only using extrapolation factors where toxicological data are lacking. For the BERA, the NOAEL TRV for DDE should be revisited. The NOAEL of 0.12 mg/kg bw/day calculated from Lincer (1975), is recommended for the BERA, because it is the highest NOAEL below the selected LOAEL based on reproduction. At this NOAEL, no effect on American kestrel reproduction was observed.

DDT
Six toxicity studies exposing birds to dietary DDT (Table 5-38) were reviewed. The lowest LOAEL, 0.15 mg/kg bw/day, calculated from Stickel and Rhodes (1970) was selected as the LOAEL TRV. A NOAEL TRV for DDT was extrapolated from the selected LOAEL TRV using a UF of 5. The resulting NOAEL TRV, 0.030 mg/kg bw/day, was selected for DDT.

There is some uncertainty associated with the selected TRVs for DDT. As in all eggshell thinning studies conducted in the laboratory where no other reproductive effects are reported, it is unknown what degree of eggshell thinning will result in adverse population level effects on reproduction. Reproductive effects on field populations of birds have been documented when eggshell thinning has reached 15 to 20% (Lincer 1975; Peakall et al. 1975; Anderson and Hickey 1972, as cited in White et al. 1984). Eggshells were only 4.6% thinner in the treated group than in the control group in the selected study, therefore, this LOAEL represents a conservative threshold value that may potentially overestimate risk to birds at the population level. There is additional uncertainty with the use of a UF to extrapolate a chronic NOAEL TRV from a chronic LOAEL TRV. For the BERA, the TRVs for DDT should be revisited.

Total DDT
Nine studies that measured the dietary toxicity of technical DDT or some other mixture of DDT and its metabolites to birds were evaluated for TRV selection (Table 5-39). The lowest LOAEL, 1.8 mg/kg bw/day, was calculated from the results of Davison and Sell (1974) and was selected as the LOAEL TRV for total DDT. The NOAEL of 0.18 mg/kg bw/day, based on the same study, was selected as the NOAEL TRV.

There is uncertainty associated with the selected TRVs for total DDT because the impact of eggshell thinning on reproductive success at a population level is unknown. Eggshell thinning was statistically different from the control group with a difference of about 6% (Davison and Sell 1974). Reproductive effects on field populations of birds have been documented when eggshell thinning has reached 15 to 20% (Lincer 1975; Peakall et al. 1975; Anderson and Hickey 1972 as cited in White et al. 1984). This selected LOAEL represents a very conservative threshold value and may overestimate risk to birds at the population level. For the BERA, the TRVs for total DDT should be revisited.

5.2.2.1.16 BEHP
Three studies on the toxic effects of dietary BEHP on birds were reviewed and evaluated for TRV selection (Table 5-40). The only calculated LOAEL of 329 mg/kg bw/day was selected as the LOAEL TRV. The reproductive NOAEL, 1.45 mg/kg/day, from Peakall (1974) was chosen as the NOAEL TRV because it was based on the same endpoint (reproduction) as the selected LOAEL.

There is a large difference between the selected NOAEL and LOAEL, and there is uncertainty associated with both of these values because the literature dataset for BEHP toxicity to birds is limited to three studies using highly variable dose concentrations.

5.2.2.1.17 Dibutyl Phthalate
No studies reporting the dietary toxicity of dibutyl phthalate to birds were identified. TRVs selected for BEHP were used as a surrogate for evaluating risks to dibutyl phthalate.

5.2.2.2 Mammal TRVs
This section presents a summary of the laboratory and field toxicity studies reviewed for Round 2 COPCs identified for mammal receptors (Table 5-5), and selected TRVs used to estimate risks. Details on the TRV selection process and on the studies reviewed are presented in the TRV Technical Memorandum and in the TRV Technical Memorandum Addendum (Attachment G5). TRVs for PCBs were selected for both total PCBs (generally based on Aroclors) and for 2,3,7,8,-TCDD (as TEQs for dioxins and furans [dioxin TEQ], and dioxin-like PCB congeners [PCB TEQ]).

A single NOAEL and LOAEL TRV were selected for each Round 2 COPC and used for all wildlife receptors (i.e., mink and river otter).

5.2.2.2.1 Dioxin TEQ and PCB TEQ

Dioxin TEQ and PCB TEQ exposure doses for each mammal receptor species were compared to the TRVs selected for 2,3,7,8-TCDD. A total of ten studies on the potential adverse effects of TCDD and dioxin-like PCB congeners on mammals were reviewed. Table 5-41. The lowest mink LOAEL of 2.24 ng/kg bw/day calculated from Tillitt et al. (1996) was selected as the LOAEL TRV. The selected mammal LOAEL TRV was selected for TEQ because it represents the most conservative value reported in the literature based on a chronic mink study where a relevant dietary fish exposure was used. A NOAEL TRV for TEQ was extrapolated from the selected LOAEL TRV using a UF of 5. The resulting NOAEL TRV, 0.44 ng/kg bw/day, was selected for TEQ.

There is high uncertainty associated with the selected TRVs for TEQ. The selected LOAEL TRV is based on a toxicological study where adverse effects on reproduction was observed in mink fed field-collected carp from Saginaw Bay, Lake Huron. Mink were fed a prepared diet containing various percentages of field-collected carp (10%, 20%, and 40%). Effects on reproduction were observed in all mink fed any percentage of the field-collected carp; therefore, this study only has quantitative relevance to mink exposed to chemical mixtures similar those found in the Saginaw Bay fish. In addition, there is uncertainty associated with these LOAELs because the field-collected carp likely contained other uncharacterized, organic chemicals that could have contributed to the reproductive toxicity reported in mink. Field-collected carp also had reported concentrations of total PCBs. In accordance with EPA's comments (EPA 2006e), the use of TRVs based on studies using field-collected fish should be revisited for use in the BERA. There is additional uncertainty with the use of a UF to extrapolate a chronic NOAEL TRV from a chronic LOAEL TRV.

5.2.2.2.2 Total PCBs

Fourteen studies on the potential adverse effects of PCBs on mammals were reviewed (Table 5-42). The lowest mink LOAEL of 0.037 mg/kg bw/day calculated from Restum et al. (1998) was selected as the LOAEL TRV. The selected LOAEL TRV represents the most conservative LOAEL reported in the literature based on a chronic mink study in which a relevant dietary fish exposure was used. A NOAEL TRV for TEQ was extrapolated from the selected LOAEL TRV using a UF of 5. The resulting NOAEL TRV, 0.0074 ng/kg bw/day, was selected for TEQ.

There is uncertainty associated with using a study relying on field-collected fish (Restum et al. 1998) to represent the NOAEL and LOAEL TRVs for total PCBs because field-collected fish likely contained other uncharacterized, organic chemicals that could have contributed to the reproductive toxicity reported in mink. In accordance with EPA's comments (EPA 2006e), the use of TRVs based on studies using field-collected fish should be revisited for use in the BERA. There is additional uncertainty with the use of a UF to extrapolate a chronic NOAEL TRV from a chronic LOAEL TRV.

5.2.2.2.3 Antimony

EPA and its partners (EPA 2006a) recommended the use of EPA Eco SSLs for those metals where SSLs are available for mammals. The bird Eco SSL for antimony, 0.59 mg/kg bw/day, was used for screening and Round 2 COPC identification (Section 5.2.1). For the purposes of risk characterization, Eco SSLs were not considered appropriate (see above for rationale), and dietary studies from the original toxicological literature were considered more representative, and therefore, more appropriate.

Only one of the five toxicological studies on oral exposure to antimony was based on dietary exposure (Table 5-43). In this study, no effect on rat growth was observed following exposure to dietary antimony (1,489 mg/kg bw/day) for 90 days. This was selected as the dietary NOAEL TRV for antimony. No LOAEL was selected because no dietary studies reporting LOAELs were available.

There is high uncertainty associated with the selected unbounded NOAEL because the literature dataset for the dietary toxicity of antimony to mammals is limited to one dietary study in which no toxicity was observed. Studies in which laboratory rodents were exposed to antimony via drinking water indicate higher toxicity (LOAELs ranged from 0.060 to 43.9 mg/kg bw/day). However, because drinking water ingestion involves a method of uptake and absorption that is different than the selected dietary pathway for wildlife receptors in the Round 2 ERA , none of the drinking water studies were considered appropriate for evaluating risk to mammal receptors through a dietary approach. Thus, the selected TRVs may underestimate oral exposure risk to mammals from antimony; however, the study used to calculate the TRVs is based on a dietary pathway, the most applicable exposure route.

5.2.2.2.4 Copper

EPA and its partners (EPA 2006a) recommended the use of EPA Eco SSLs for those metals where SSLs are available for mammals. The bird Eco SSL for copper, 5.82 mg/kg bw/day, was used for screening and Round 2 COPC identification (Section 5.2.1). For the purposes of risk characterization, Eco SSLs were not considered appropriate (see above for rationale), and dietary studies from the original toxicological literature were considered more representative, and therefore, more appropriate.

Three studies on the toxic effects of dietary copper to mammals were reviewed and evaluated for TRV selection (Table 5-44). The LOAEL and NOAEL of 26 and 18 mg/kg bw/day copper derived from Aulerich et al. (1982) were chosen as the TRVs for copper. The selected TRVs for copper in mammals represents concentrations protective of the most sensitive mammal species measured in the literature review (mink), which is also a selected receptor species for Portland Harbor.

There is some uncertainty associated with the selected TRVs because the literature dataset for the dietary toxicity of copper to mammals is limited to three studies.

5.2.2.2.5 Lead

EPA and its partners (EPA 2006a) recommended the use of EPA Eco SSLs for those metals where SSLs are available for mammals. The bird Eco SSL for lead, 4.7 mg/kg bw/day, was used for screening and Round 2 COPC identification (Section 5.2.1). For the purposes of risk characterization, Eco SSLs were not considered appropriate (see above for rationale), and dietary studies from the original toxicological literature were considered more representative, and therefore, more appropriate.

Only one of the eleven toxicological studies on oral exposure to lead was based on dietary exposure (Table 5-45). In this study (Azar et al. 1973), rats were exposed to dietary lead for 2 years, and the effect on reproduction was measured. The LOAEL and NOAEL of 90 and 11 mg/kg bw/day, respectively, were selected as the lead TRVs for mammals.

There is some uncertainty associated with the selected TRVs because the literature dataset for the dietary toxicity of lead is limited to one study only. The selected dietary LOAEL and NOAEL are higher than the effect levels reported in the studies that used drinking water, though drinking water effect levels were widely variable (LOAELs ranged from 3.3 to 120 mg/kg bw/day lead for growth and reproduction). However, because drinking water ingestion involves a method of uptake and absorption that is different than the selected dietary pathway for wildlife receptors in the Round 2 ERA, none of the drinking water studies were considered appropriate for evaluating risk to mammal receptors through a dietary approach. LOAELs calculated from gavage studies are also lower (LOAELs range from 1.5 to 5.5 mg/kg bw/day for behavior and reproduction). Thus, the selected TRVs may underestimate oral exposure risk to mammals from lead; however, the study used to calculate the TRVs is based on a dietary pathway, the most applicable exposure route.

5.2.2.2.6 Mercury

Seven toxicity studies on dietary mercury using mink and laboratory rodents were evaluated for TRV selection (Table 5-46). The LOAEL and NOAEL of 0.07 and 0.02 mg/kg bw/day, respectively, were selected as the TRVs for mercury. These were the lowest TRVs calculated from the toxicological literature for mink. A lower LOAEL (0.0084 mg/kg bw/day) was reported in Verschuuren et al. (1976); however, this study was not used to derive the selected TRV because low-effect doses were reported in Dansereau et al. (1999), in which mink, one of the selected mammal receptors in Portland Harbor, were exposed over a chronic duration for multiple generations through an ecologically relevant exposure (via partial fish consumption). The NOAEL was the lowest NOAEL calculated from the reviewed studies.

There is uncertainty associated with the selected TRVs because the field-collected fish (making up 40% of the prepared diet) may have contained other, uncharacterized organic chemicals that could have contributed to the reproductive toxicity reported in mink. In accordance with EPA's comments(EPA 2006e), the use of TRVs based on studies using field-collected fish should be revisited for use in the BERA.

5.2.2.2.7 Selenium

Six laboratory studies on the toxicity of selenium to mammals were evaluated for TRV selection (Table 5-47). The LOAEL and NOAEL of 0.08 and 0.055 mg/kg bw/day, respectively, were selected as the TRVs for selenium. These were the lowest TRVs calculated in the literature reviewed.

There is some uncertainty associated with the selected TRVs because the study was conducted over a subchronic (six-week) period. However, no dietary study reviewed was conducted over a chronic period (i.e., one year) or during a critical life stage. Ferm et al. (1990) exposed hamsters via oral gavage during a critical life stage (gestation) and reported much higher LOAELs (ranging from 7.1 to 8.7 mg/kg bw/day selenium). Thus, the selected TRVs may overestimate chronic risk to mammals exposed to selenium.

5.2.2.2.8 PAHs: Total PAHs

Three studies on the toxicity of benzo(a)pyrene to mammals were evaluated for TRV selection (Table 5-48). No studies were found on the toxicity of a PAH mixture to mammals, and benzo(a)pyrene toxicity studies were used as a surrogate to evaluate total PAHs. The LOAEL of 10 mg/kg bw/day reported in MacKenzie and Angevine (1981) was selected as the LOAEL TRV. This was the only LOAEL reported in the reviewed literature, and this study was conducted over a critical life stage (during pregnancy). A NOAEL TRV was extrapolated from the selected LOAEL TRV using a UF of 5. The resulting NOAEL TRV, 2.0 mg/kg bw/day, was selected.

There is high uncertainty associated with the selected LOAEL using benzo(a)pyrene toxicity as a surrogate for total PAH toxicity, which is a highly conservative approach. There is further uncertainty because gavage is not a preferred exposure route; however, no LOAELs were reported in the two dietary studies that were reviewed. In addition, the selected TRVs because the literature dataset for benzo(a)pyrene toxicity in mammals is limited to three studies using laboratory mice, and only one study reports an effect-level concentration and use of a UF to extrapolate a chronic NOAEL TRV from a chronic LOAEL TRV is uncertain.

5.2.2.2.9 Total DDTs

Numerous toxicity studies on DDT in mammals were reviewed. A total of 21 studies on the potential adverse effects of DDT and its metabolites to mammals were reviewed. Investigations that addressed separate exposures to isomers of DDT, DDD, and DDE were found, but insufficient data were available from the literature to derive separate TRVs for DDD, DDE, and DDT. In addition, the only studies that reported adverse effects for these compounds (DDD, DDE, and DDT) did not conduct statistical analyses of the results. Therefore, TRVs were derived only for total DDTs based on studies that used any mixture or technical grade of DDT. All studies reviewed on the oral toxicity of DDT and its metabolites are presented in the following subsections.

Fifteen studies that measured the toxicity of technical DDT or some other mixture of DDT and its metabolites to mammals were evaluated for TRV selection (Table 5-49). The lowest LOAEL of 1.3 mg/kg bw/day, as calculated from Ware and Good (1967) was selected as the LOAEL for total DDT. The selected LOAEL represents the most conservative effect-level concentration reported in all of the literature reviewed. Based on EPA's comments to the Ecological PRE (EPA 2006e), a selected NOAEL TRV for total DDTs (0.26 mg/kg bw/day) was extrapolated from the selected LOAEL TRV using a UF of 5.

LWG recommends the use of actual toxicity data and only using extrapolation factors where toxicological data are lacking. For the BERA, the NOAEL TRV for DDE should be revisited. The NOAEL of 1.2 mg/kg bw/day calculated from Duby et al. (1971), is recommended for the BERA, because it is the highest NOAEL below the selected LOAEL based on reproduction. At this NOAEL, no effect on rat reproduction was observed following exposure to dietary DDTs for multiple generations.

5.3 BIRD EGG ASSESSMENT

As an additional line of evidence for evaluating risks to piscivorous birds (i.e., osprey and bald eagle), risks to specific COIs were evaluated by estimating concentrations in egg tissues and comparing estimated concentrations to bird egg TRVs from the literature. Egg tissue concentrations were estimated using prey tissue concentrations and prey tissue-to-egg tissue (BMFs. This line of evidence was evaluated to be protective of developing embryos and a sensitive life-stage.

Chemical concentrations in bird eggs are not available from the Study Area; however, bird egg tissue-residue concentrations were estimated using the following equation:

$$EPC_{egg} = BMF \times C_{tiss} \qquad \textbf{Equation 5-9}$$

Where:

C_{egg} = estimated exposure concentration in bird egg tissue (mg/kg ww)
BMF = biomagnification factor
C_{tiss} = tissue concentration in fish prey item (mg/kg ww)

BMFs were specific to each chemical, and were selected from region-specific literature. All Round 2 COPC/receptor pairs identified for the bird egg tissue line of evidence in (see Section 5.1.5) were evaluated for the tissue assessment line of evidence.

5.3.1 Round 2 COPC Exposure Assessment

Round 2 COPC exposure estimates were expressed as a bird egg tissue concentration, representing prey tissue ingestion and biomagnification of the chemical concentrations from fish prey to bird egg tissues. Bird egg tissue concentrations were

estimated using Equation 5-9. BMFs were derived from the literature (Section 5.3.2.2). The chemical concentration in osprey and bald eagle fish prey (C_{prey}) was calculated from Round 2 COPC concentrations in each component of the receptor's diet and estimated of each component's fraction of the total diet.

For example, the concentrations in food for a piscivorous bird that might ingest both two species of fish prey would be estimated as follows:

$$C_{prey} = (C_1 \times F_1) + (C_2 \times F_2)$$ **Equation 5-10**

Where:

C_{prey}	=	concentration in prey items (mg/kg ww)
C_1	=	concentration in fish prey tissue prey item 1 (mg/kg ww)
F_1	=	fraction of the receptor diet consisting of fish prey item 1 (kg fish/kg food)
C_2	=	concentration in fish prey tissue item 2 (mg/kg ww)
F_2	=	fraction of the receptor diet consisting of fish prey item 2 (kg fish/kg food)

The same dietary fractions of prey used to estimate bird egg tissues for bald eagle and osprey are the same dietary fractions used to estimated dietary exposure doses for these receptors in the dietary dose approach (Section 5.2.2; Table 5-7). The dietary fraction of each component in each receptor's diet was based on information from the literature. Prey items for piscivorous birds included fish species (i.e., largescale sucker, carp, peamouth, smallmouth bass, northern pikeminnow, black crappie, and brown bullhead) collected in the field within the Study Area. Not all Round 2 COPCs were analyzed in all prey items. Therefore, surrogate species were selected from similar feeding guilds to represent prey tissue concentrations. These surrogate species for specific Round 2 COPCs are noted in Table 5-7.

5.3.1.1 BMFs

Literature-based BMFs were used to model the concentration of selected chemicals in developing embryos of piscivorous birds. Several literature sources were reviewed that measured the relationship between concentrations in prey fish tissue and the tissue concentrations in piscivorous bird eggs. BMFs were compiled for dioxins, PCBs, DDE, and mercury using available literature for bald eagle, osprey, blue heron, herring gulls, and brown pelican. BMFs were based on seven studies where data were collected in the Willamette River (Henny et al. 2003; Thomas and Anthony 1999), the Lower Columbia River (Buck 2004; Thomas and Anthony 1999), Great Lakes region (Braune and Norstrom 1989; Giesy et al. 1995; Kubiak and Best 1991, as cited in Giesy et al. 1995); and South Carolina (Blus et al. 1977). Table 5-50 presents a summary of all the BMFs compiled from the reviewed literature sources.

In accordance with EPA's comments to the Ecological PRE (EPA 2006b), BMFs were selected from the reviewed literature to be most representative of the Study Area

region (from the Willamette River) and selected piscivorous bird receptor species (i.e., bald eagle and osprey). Therefore, in accordance with EPA comments, the BMFs for all Round 2 COPCs (except mercury) were based on osprey data from the Willamette River reported by Henny et al. (2003). For mercury, no Willamette-specific data were available, and the selected BMF was based on bald eagle data from the Lower Columbia River (Buck 2004). Table 5-51 presents the selected BMFs used to estimate bird egg tissue concentrations.

5.3.1.2 Estimated Bird Egg Tissue Concentrations

Bird egg tissue concentrations were calculated for each bald eagle and osprey based on the information presented in preceding sections. Round 2 COPC concentrations in prey tissue (C_{prey}) that were used in Equations 5-9 to estimate bird egg tissue concentrations for bald eagle and osprey were based on an estimate of each Round 2 COPC EPCs in each of the various prey tissue items. EPCs were calculated for all Round 2 COPCs for each of the prey tissue items as an upper bound estimate (UCL). All wildlife receptors were assumed to forage in an area that includes the entire Study Area. Thus, for each prey type ingested by those species, the EPC was calculated using all data from the Study Area as one exposure dataset. These tissue EPCs based on UCLs are presented in Table 5-10. Estimated bird egg tissue concentrations for bald eagle and osprey are presented in Table 5-52.

5.3.2 Round 2 COPC Effects Assessment

This section presents a brief summary of the toxicity literature for each of the Round 2 COPCs and the selected TRVs for bird egg Round 2 COPCs. NOAEL and LOAEL TRVs were presented in the Ecological PRE (Windward 2005) based on an extensive search on the available toxicological literature for the development of bird and mammalian wildlife dietary TRVs and were selected to represent conservative screening level toxicological thresholds. EPA's comments to the Ecological PRE state that "[bird egg] TRVs derived for non-domestic species more specific to the Willamette River should be used for the next phase of the risk assessment…" (EPA 2006b). EPA further clarified that the selected screening level TRVs for bird eggs presented in the Ecological PRE may be overly conservative for bird receptors in the Study Area (EPA 2006a). Thus, changes to bird egg TRVs (to be based on Willamette-specific species) based on EPA comments to TRVs (EPA 2006a, b) were adopted for risk characterization of Round 2 COPCs in this Round 2 ERA and incorporated into the selected of NOAEL and LOAEL TRVs. The literature search and guidelines for TRV selection for bird eggs are described in detail in the TRV Technical Memorandum. Toxicological data presented in this section are assessed in combination with exposure data (presented in Section 5.3.1) in the risk characterization (Section 5.4).

It should be noted that selected TRVs may not be appropriate for use in the BERA may be revisited prior to the preparation of the BERA.

5.3.2.1 Dioxin TEQ and PCB TEQ

Seventeen studies were reviewed on egg tissue residues of dioxins and/or dioxin-like chemicals associated with no effect and adverse effect levels (Table 5-53). Both egg injection studies and field studies reporting egg residues and associated effects on embryo survival and development were reviewed. These endpoints (embryo survival and development) appear to be the most sensitive effects resulting from dioxins and dioxin-like exposure seen in birds, and are clearly related to reproductive success in avian populations (EPA 2003a; Hoffman et al. 1996).

The selected LOAEL TRV for TEQ was based on the lowest LOAEL TRV reported for a representative piscivorous bird species from the Willamette River. At 31.98 pg/g ww of 2,3,7,8-TCDD, reduced productive and eggshell thinning were observed in bald eagles nesting in the Columbia River estuary from 1980 to 1987 (Anthony et al. 1993). This LOAEL TRV was selected. The highest NOAEL below this selected LOAEL based on a representative Willamette bird species was selected. At 2.3 pg/g ww, Henny et al. (2003) reported no effect on osprey productivity in the Willamette River in 1993.

There is uncertainty associated with using LOAEL TRVs based on field-collected data. Bird egg tissues in the field contained other uncharacterized, organic chemicals (e.g., PCBs, DDE) that could have contributed to the reproductive toxicity reported in bald eagles. However, field-collected data allow for the selection of Willamette-specific data to derive toxicological data to protect Willamette receptors. In accordance with EPA's comments (EPA 2006e), the use of TRVs based on studies using field-collected fish should be revisited for use in the BERA.

5.3.2.2 PCBs

Twenty-seven laboratory studies and field studies reporting PCB concentrations and associated effects were reviewed (Table 5-54). The most sensitive endpoint for PCB toxicity in birds is reproductive impairment associated with egg residues (Hoffman et al. 1996). The selected LOAEL TRV for total PCBs was based on the lowest LOAEL TRV reported for a representative piscivorous bird species from the Willamette River. Bald eagle egg tissues as low as 4.5 µg/g ww were associated with impaired five year productivity in bird populations from 14 states in the United States (Wiemeyer et al. 1984). No effect on five-year productivity was reported with bald eagle egg tissues as low as 3.0 µg/g ww was measured (Wiemeyer et al. 1993). The NOAEL and LOAEL TRVs of 4.5 and 3.5 µg/g ww, were selected as the bird egg TRVs for PCBs.

There is general uncertainty associated with the egg PCB LOAELs reported in the field literature, because concentrations of PCBs are frequently closely correlated with concentrations of DDE (Giesy et al. 1995), and it is difficult to separate the adverse reproductive effects of PCBs from those of DDE (Wiemeyer et al. 1984). Nearly all field studies reviewed reported levels of both PCBs and DDE associated with adverse effects on bird populations. However, field-collected data allow for the selection of Willamette-specific data to derive toxicological data to protect Willamette receptors.

In accordance with EPA's comments(EPA 2006e), the use of TRVs based on studies using field-collected fish should be revisited for use in the BERA.

5.3.2.3 DDE

A total of 31 laboratory and field studies reporting DDE or DDT concentrations in eggs and associated reproductive effects were reviewed and considered for TRV selection (Table 5-55). Egg TRVs were selected for DDE only because birds appear to be most sensitive to DDE. The selected LOAEL TRV of 3.5 mg/kg ww DDE was based on Wiemeyer et al. (1984). At this egg concentration (reported in field-collected bald eagle eggs), offspring productivity was below the level necessary to maintain a stable population (Wiemeyer et al. 1984). Thus, the selected LOAEL of 3.5 mg/kg ww DDE represents a conservative effect threshold based on the most sensitive bird species measured in the literature reviewed. Wiemeyer et al. (1984) also reported a NOAEL of 1.3 µg/g ww. This NOAEL was selected as the NOAEL TRV for DDE in bird eggs.

There is uncertainty associated with using LOAEL TRVs based on field-collected data. Bird egg tissues in the field contained other uncharacterized, organic chemicals (e.g., dioxins, PCBs) that could have contributed to the reproductive toxicity reported in bald eagles. However, field-collected data allow for the selection of Willamette-specific data to derive toxicological data to protect Willamette receptors. In accordance with EPA's comments (EPA 2006e), the use of TRVs based on studies using field-collected fish should be revisited for use in the BERA.

5.3.2.4 Mercury

A total of eight laboratory studies and field studies reporting mercury concentrations and associated effects were reviewed (Table 5-56). No data on Willamette-specific species were available from the reviewed toxicological studies. Furthermore, no LOAELs were reported in the field studies reviewed. Therefore, the lowest acceptable LOAEL derived from the laboratory studies was 0.74 µg/g ww mercury (Heinz and Hoffman 2003), and this was selected as the LOAEL TRV for mercury in bird eggs. The highest NOAELs reported below the selected LOAEL were reported in the two field studies reviewed. A NOAEL of 0.5 µg/g ww was selected, as recommended by Wiemeyer et al. (1984), where adverse effects on bald eagle productivity were not expected. This NOAEL is consistent with the field NOAELs reported in Haseltine et al. (Haseltine et al. 1981), in which hatching success was unaffected in red-breasted merganser eggs with concentrations ranging from 0.51 to 0.52 µg/g ww.

There is uncertainty associated with the selected egg LOAEL because the mercury egg concentrations reported in the laboratory varied widely across the toxicological studies reviewed. There are uncertainties associated with the selected NOAEL TRV because it is difficult to establish threshold levels for mercury in birds based on field-collected eggs (Giesy et al. 1995) because mercury concentrations in eggs are often associated with other organochlorine concentrations such as DDE and PCBs (Eisler 1987; Haseltine et al. 1981; Wiemeyer et al. 1984), and the interaction may intensify

adverse effects to avian populations. However, field-collected data allow for the selection of Willamette-specific data to derive toxicological data to protect Willamette receptors. In accordance with EPA's comments (EPA 2006e), the use of TRVs based on studies using field-collected fish should be revisited for use in the BERA.

5.4 ROUND 2 RISK RESULTS

This section presents risk estimates, for all wildlife receptors across both the dietary dose and bird egg lines of evidence based on the assumptions presented for each line of evidence evaluated (as presented in Sections 5.2 and 5.3 for the dietary dose and bird egg lines of evidence, respectively).

HQs were derived for each Round 2 COPC/receptor pair using the following equation:

$$HQ = \frac{Dose}{TRV} \quad \text{or} \quad HQ = \frac{EPC_{egg}}{TRV}$$ **Equation 5-10**

Where:

HQ	=	ecological hazard quotient (unitless)
Dose	=	estimated dietary exposure dose (mg/kg bw/day)
EPC_{egg}	=	estimated egg exposure concentration (mg/kg ww)
TRV	=	toxicity reference value (mg/kg bw/day or mg/kg ww; based on dietary dose or bird egg NOAEL and LOAEL TRV)

HQ results are presented for each line of evidence. The uncertainty section (Section 5.4.2) evaluates the uncertainty around the assumptions and methods for deriving the HQs presented in this section (Section 5.4.1) to arrive at general risk conclusions for wildlife receptors and which Round 2 COPCs are identified as Round 2 iCOCs. Risk conclusions and wildlife iCOCs are identified in Section 5.4.3.

5.4.1 Dietary Dose Assessment

For spotted sandpiper, when risks were evaluated using all beaches, four Round 2 COPCs (dioxin TEQ, PCB TEQ, total PCBs, and sum DDD) had both NOAEL and LOAEL HQs > 1.0 for clam and/or worms, and eight Round 2 COPCs (arsenic, chromium, lead, mercury, aldrin, sum DDE, sum DDT and total DDTs) had NOAEL HQs > 1.0 for clams and/or worms (Table 5-57). All other Round 2 COPCs had HQs < 1.0.

When shorebird risk was assessed for each habitat beach area, 15 Round 2 COPCs had worm or clam NOAEL and/or LOAEL HQs > 1.0 in at least one shorebird beach area (Table 5-58). Table 5-3 shows those shorebird beaches with LOAEL HQs > 1.0. The following Round 2 COPC HQ results were determined on a beach by beach basis:

- For total PCBs and PCB TEQ, both NOAEL and LOAEL HQs were > 1.0 at several shorebird beaches (Figure 5-3). LOAEL HQs based on 100% worm diet were > 1 for total PCBs at B3, B15, B16, B22, B24, and B27, and LOAEL HQ were > 1.0 based on 100% clam diet at only one beach, B15. PCB TEQ HQs were similar; LOAEL HQs were > 1.0 based on the worm diet at B3, B16, and B22 and LOAEL HQs for PCB TEQ were < 1.0 at all beaches based on the clam diet.
- For dioxin TEQs, both NOAEL and LOAEL HQs were > 1.0 based on the worm diet at two beaches (B16 and B17; Figure 5-3), and NOAEL HQs were > 1.0 at four other beaches. For shorebird modeled using the clam diet, only two beaches has NOAELs > 1 and all other HQs were < 1.0.
- For sum DDD, both NOAEL and LOAEL HQs were > 1.0 at two shorebird beaches (Figure 5-3) based on the worm diet (B16 and B17). No LOAEL HQs were > 1.0 for sum DDE, sum DDT, and total DDT; however, NOAEL HQs were > 1.0 for all or some of these Round 2 COPCs at three beach locations (B16, B17, and B22).
- For aldrin, NOAEL and LOAEL HQs were > 1.0 for worms at one beach (B22; Figure 5-3). Aldrin HQs were < 1.0 at all other beaches.
- For zinc, NOAEL HQs were > 1.0 for clams at one beach (B20) and < 1.0 at all other beaches.
- The chromium, lead, and mercury, NOAEL HQs were > 1.0 and < 5.0 based on both the worm and clam diets at two or more or all shorebird beach areas. All beach areas had LOAEL HQs < 1.0 for these metals.
- For arsenic, NOAEL HQs were > 1.0 at two beaches (B6 and B18) based on the worm diet, and < 1.0 for all beaches based on the clam diet. All beach areas had LOAEL HQs < 1.0 for arsenic.
- For all other Round 2 COPCs (TBT, cadmium, copper, total PAHs, BEHP), NOAEL and LOAEL HQs were < 1.0 at all shorebird beaches evaluated.

Using the dietary dose line of evidence, nine Round 2 COPCs had HQs > 1.0 for the five wildlife receptors that were evaluated using a Study Area-wide exposure assumption only (i.e., hooded merganser, osprey, bald eagle, mink and river otter; Tables 5-59 through 5-63).

- For hooded merganser, nine Round 2 COPCs (dioxin TEQ, PCB TEQ, total PCBs, mercury, zinc, sum DDE, sum DDT,

total DDTs, and BEHP) exceeded a NOAEL TRV, with NOAEL HQs ranging from 1.0 to 5.0. No Round 2 COPCs had LOAEL HQs > 1.0.

- For bald eagle, one Round 2 COPC (total PCBs) exceeded a NOAEL TRV, with a NOAEL HQ of 1.5. No Round 2 COPCs had LOAEL HQs > 1.0.
- For osprey, four Round 2 COPCs (total PCBs, mercury, sum DDT, and BEHP) exceeded a NOAEL TRV, with NOAEL HQs ranging from 1.0 to 3.0. No Round 2 COPCs had LOAEL HQs > 1.0.
- For mink, three Round 2 COPCs exceeded both NOAEL and LOAEL TRVs: dioxin TEQ, PCB TEQ, and total PCBs. LOAEL HQs ranged from 1.0 to 10. No other Round 2 COPCs had HQs > 1.0.
- For river otter, two Round 2 COPCs (PCB TEQ and total PCBs) exceeded both a LOAEL and NOAEL TRVs. One Round 2 COPC (dioxin TEQ) exceeded a NOAEL TRV (NOAEL HQ = 4.7).. No other Round 2 COPCs had HQs > 1.0.

5.4.2 Bird Egg Assessment

Using the bird egg approach line of evidence, four Round 2 COPCs had HQs > 1.0 for osprey and bald eagle: Dioxin TEQ, PCB TEQ, total PCBs, and sum DDE. LOAEL HQs ranged from 1.0 to 10 for these four Round 2 COPCs. Table 5-64 presents the HQs for all bird egg Round 2 COPCs.

5.5 UNCERTAINTIES

This section presents the uncertainties associated with the assumptions and methods for deriving HQs for all wildlife Round 2 COPC/receptor pairs, as presented in Section 5.4.1. Uncertainties are presented in the following subsections based on the dietary dose assessment (Section 5.4.2.1), and the bird egg assessment (Section 5.4.2.2). A summary of all of the uncertainties associated with each Round 2 COPC is presented in Section 5.4.2.3.

5.5.1 Dietary Dose Assessment

There are uncertainties associated with the effects and exposure assessment of the dietary dose LOE. The primary uncertainties associated with the effects assessment of the dietary dose approach are: 1) the selected TRVs, 2) the lack of TRVs for COIs with no toxicological literature, and 3) the TEQ approach for dioxins and dioxin-like PCB congeners. The uncertainties associated with the exposure assumptions for wildlife receptors are the assumption that: 1) dietary prey assumptions reflect the portions of prey consumed by receptors, 2) surface water ingestion is a minor

pathway for wildlife Round 2 COPC exposure, 3) that all wildlife receptors, with the exception of spotted sandpiper, would forage in sediments and tissues throughout the 9-mile Study Area and ingest sediments from the 0-to-20-ft water depth, 4) mink may ingest up to 9% sediment while feeding 5) the assumption that site use factors are equal to 1.0 for all wildlife receptors, and 6) that prey tissue and sediment EPCs used to estimate dietary doses are represented by UCL concentrations. For spotted sandpiper, there are uncertainties associated with the exposure assumptions that 1) spotted sandpipers would utilize only the habitat within each shorebird beach area identified for breeding and forage only within the habitat area, and 2) the use of laboratory-exposed worms and estimate benthic tissue where no tissue data are available to represent prey tissue for shorebirds. Each of these uncertainties are discussed and evaluated in the following subsections.

5.5.1.1 Selected TRVs

There are uncertainties associated with the selected bird and mammal dietary dose TRVs. In general, the uncertainties include the following:

- Few of the laboratory toxicological studies used to derive TRVs were conducted using relevant bird or mammal species present in the Willamette River.
- The laboratory studies on which TRVs are based were conducted in controlled settings using single-contaminant exposures. Effects associated with multiple-chemical exposure and other environmental stressors present at the Study Area (e.g., habitat loss) were not factored into these studies. It is unknown if these factors would result in additive, synergistic, antagonistic, or neutral effects on overall risk conclusions.
- Limited toxicological data were available for some Round 2 COPCs and selected TRVs for some Round 2 COPCs were based on a limited dataset. For some Round 2 COPCs, studies where birds or mammals were dosed using non-preferred exposure conditions (e.g., weekly injections) were the only toxicological data available, and these TRVs were selected based on non-preferred exposure conditions.
- NOAELs were extrapolated from chronic LOAELs using UFs, when no NOAEL was available from some Round 2 COPCs, and chronic NOAELs were extrapolated from subchronic NOAELs when no chronic NOAEL was available.
- No LOAEL was available for total PAHs for birds, so only a NOAEL could be evaluated for total PAHs.

The uncertainties of the selected bird and mammal dietary dose TRVs for each Round 2 COPC are summarized in Table 5-73 through 5-78.

5.5.1.2 COIs with No TRVs

Of the COIs identified, ten were not included in the Round 2 COPC screen for birds because no data were available for the development of TRVs. No toxicological studies identified for antimony, silver, 2-methylnaphthalene, hexachloroethane, 2-methylphenol, 4-methylphenol, phenol, benzyl alcohol, dibenzofuran, and N-nitrosodiphenylamine. The risk associated with these chemicals to bird receptors is unknown because no toxicological data are available.

For mammals, six COIs were not included in the Round 2 COPC screen because no TRV data were available. No toxicological studies identified for silver, 2-methylphenol, 4-methylphenol, benzyl alcohol, dibenzofuran, and N-nitrosodiphenylamine. The risk associated with these chemicals to bird receptors is unknown because no toxicological data are available.

5.5.1.3 TEQ Approach

For the calculation of dioxin and dioxin-like PCB congener risks using the TEQ approach, fish, benthic invertebrate, and sediment samples were analyzed for dioxins, furans, and PCB congeners, and TEFs were used to account for toxicity relative to 2,3,7,8-TCDD. The TEFs used to calculate TEQs for dioxin-like PCB congeners were WHO consensus values for birds and mammals from Van den Berg et al. (1998); these TEFs are presented in Appendix G1. The rationale for the use of TEFs is based on evidence that there is a common mechanism of toxicity for certain dioxins, furans, and PCB congeners, which involves binding to the aryl hydrocarbon (Ah) receptor as an initial step. Data on the relative binding affinity of particular PCB congeners compared to 2,3,7,8-TCDD are available from in vivo and in vitro studies. These data have been used to derive TEFs for PCB congeners that show structural similarity to dioxins, bind to the Ah receptor, and elicit dioxin-specific biochemical and toxic responses.

A key uncertainty in the TEQ approach is related to the derivation of consensus TEF values. Limitations in the underlying data used to derive TEFs, such as the relevance of the endpoints in the studies and the lack of information on interspecies variability, contribute to the uncertainty.

The four most potent Ah receptor agonists in birds among PCB congeners are the non-ortho PCBs 77, 81, 126, and 169. The variability in the bird TEFs appears high for PCB congeners that have been tested on multiple species (Van den Berg et al. 1998). For PCB 77, five studies have been conducted, resulting in a TEF range of < 0.0003 to 0.15 for the various bird species tested for ethoxyresorufin-O-deethylase (EROD) induction or in ovo effects. For PCB 81, two identified studies tested several species for EROD induction, with TEFs highly variable, ranging from 0.001 to 0.5. For PCB 126 and 169, data are available from only one study (in ovo with chickens). The relevance of TEFs derived by EROD induction or in ovo studies to risk assessment based on dietary exposure is also uncertain. It is not known if the uncertainties in the bird TEFs would overestimate or underestimate risk.

The TEFs for mammals were derived from a large number of studies, with priority given to in vivo toxicity data over in vitro data. Despite the numerous biological variables such as species, strain, sex, and age included in these studies, the TEF values for a given congener generally fall within a range of about an order of magnitude for mammals (Sanderson and Van den Berg 1999). It is not known if the uncertainties in these TEFs would overestimate or underestimate risk.

Dioxin TEQs and PCB TEQ risks were calculated for mink and river otter using the updated WHO mammal TEFs (Van den Berg et al. 2006) to determine whether the use of the updated TEFs would change risk conclusions (Table 5-65). Dioxin TEQ HQs are reduced slightly using the updated mammal TEFs; however, NOAEL and LOAEL HQs are still exceeded for mink and NOAEL HQs are still exceeded by river otter. Using the updated TEFs for PCB TEQ results in lower HQs for mink and river otter, and would reduce LOAEL HQs for both mink and river otter to < 1.0.

5.5.1.4 Dietary Prey Assumptions

This section presents the uncertainties associated with the assumed dietary fractions used to derive dietary dose HQs for hooded merganser, osprey, bald eagle, mink, and river otter.

In order to develop dietary dose risk calculations, dietary fractions of prey items used were assigned for each wildlife receptor (Table 5-7) based on literature searches and best professional judgment, as presented in Section 5.2.2. There is some uncertainty associated with the dietary fractions of prey items assigned to each wildlife receptor (i.e., whether dietary fractions of each prey item should be greater or less than the fraction assigned for estimating risk calculations). Therefore, the impact of changing dietary portion for each prey item on risk estimates (NOAEL and LOAEL HQs presented in Tables 5-59 through 5-63) was evaluated for each wildlife receptor with multiple prey items (i.e., hooded merganser, bald eagle, osprey, mink, and river otter).

To evaluate the impact of varying prey portions on the NOAEL and LOAEL HQs presented in Section 5.4.1 and Tables 5-59 through 5-63, an initial calculation was conducted for each of the wildlife receptors where it was assumed each receptor consumed the only prey items with the highest tissue EPC[51] for a given chemical. For the chemical/receptor combinations presented in Table 5-66, using 100% ingestion of the prey items with the highest tissue EPC: 1) resulted in a new NOAEL or LOAEL HQ > 1.0, or 2) caused a NOAEL or LOAEL HQ exceedance to double in magnitude. For these receptors-chemical pairs, further evaluation was conducted and is described below. For all other chemical/receptor pairs, varying the prey portions resulted in similar NOAEL and LOAEL HQs, and therefore the dietary prey assumptions were assumed to be adequately conservative of Round 2 COPC/receptor pairs in the Study Area.

[51] EPC based on ProUCL generated value (see Table 5-10).

For those Round 2COPC/receptor pairs presented in Table 5-66, dietary prey portions were manipulated to determine the percent prey item with the highest prey concentration that resulted in an HQ > 1.0 or resulted in an HQ exceedance doubling in magnitude. Dietary fractions were altered to increase the consumption of the prey item with the maximum tissue EPC of a given chemical and lower the prey items with the lowest concentrations. Table 5-66 presents the results of this uncertainty analysis. For each chemical/receptor combination, a "likelihood of occurrence" was assigned as high, moderate, or low. If the dietary fraction required only a small change for a new exceedance and/or there was high uncertainty on prey portions based on the literature (i.e., no region specific studies were available), it was considered to have a high or moderate likelihood of occurrence. However, if the dietary fraction requires a significant change to change HQs and the change was not realistic based on the dietary data provided in the reviewed literature, the assumed dietary prey portions were considered adequately conservative enough for the protection of wildlife receptors in the Study Area.

Overall, changes to dietary prey portions that were significant enough to cause change to HQs were considered unlikely to occur ("low" likelihood of occurrence) in the Study Area except for several chemical/receptor combinations that would effect changes to NOAELs. A "high" likelihood of prey item portions reflecting actual exposure in the Study Area was identified for mercury/bald eagle and mercury/mink.

- For bald eagle, a small increase (1%) in the consumption of northern pikeminnow (from 5% to 6%) would results in a NOAEL HQ > 1.0. Therefore, the HQ results of mercury and bald eagle are uncertain.
- For mink, a small increase (7%) in the consumption of northern pikeminnow consumption (from 5% to 12%) would result in NOAEL HQ > 1.0 for mercury. Therefore, the HQ results of mercury and mink are uncertain.

A "moderate" likelihood of prey item portions reflecting actual exposure in the Study Area were identified for dioxin TEQ/river otter, PCBs/bald eagle, PCBs/mink, PCBs/river otter, and sum DDT/bald eagle.

- For river otters, an increase in the consumption of crayfish from 10% to 19% results in LOAEL HQ > 1.0 for dioxin TEQ. However, because mink already have a LOAEL HQ > 1.0 for dioxin TEQ, this uncertainty with the dietary assumptions would not change the risk conclusions regarding dioxin TEQ risks for carnivorous mammals in the Study Area.
- For bald eagles, a LOAEL HQ > 1.0 for total PCBs would occur if carp consumption increased from 45% to 67% of the diet.

- For mink and river otter, the NOAEL and LOAEL HQ for total PCBs doubled when carp consumption was increased from 15% to 46% and from 26% to 73%, respectively. For both mink and river otters, carp are an important source of food, and the uncertainty surrounding the dietary compositions for these species makes these changes possible, but not highly likely. Since the LOAEL HQs, the uncertainty of the dietary parameters only affects the magnitude of HQs for mammals and total PCBs.
- For bald eagle, an increase in the consumption of largescale sucker from 45% to 85% results in a NOAEL HQ > 1.0 for sum DDT. While largescale suckers are an important part of the bald eagle diet, it is uncertain whether the population of largescale suckers in the Study Area is sufficient for this to occur.

5.5.1.5 Surface Water Ingestion

The impact of surface water ingestion on the overall dietary dose exposure estimates was quantitatively evaluated to ensure that risks via water ingestion for wildlife were minor and insignificant. In order to determine the effect of water ingestion on the overall risk for wildlife receptors, a water ingestion rate was calculated using Equation 5-11 for bird receptors and Equation 5-12 for mammal receptors:

$$\mathrm{WIR} = 0.059 \times \mathrm{BW}^{0.67} \qquad \textbf{Equation 5-11}$$

$$\mathrm{WIR} = 0.099 \times \mathrm{BW}^{0.90} \qquad \textbf{Equation 5-12}$$

Where:

WIR = water ingestion rate (L/day)
BW = body weight (kg)

In order to determine the maximum possible exposure based on water ingestion, the maximum Study Area-wide water concentration was used for each chemical. However, due to the relatively low surface water concentrations, as compared to concentrations found in prey tissue, the contribution of risk from water ingestion was relatively low. Table 5-67 shows the average percent contributions from prey tissue, sediment, and surface water for each wildlife receptor.

The average percent risk from surface water ingestion was < 0.1% for all wildlife receptors. Additionally, the maximum contribution of water ingestion to the overall risk did not exceed 0.3% for any chemical/receptor combination, and can thus be said to have no significant impact on the overall risk. This quantitative analysis demonstrates that exposure to chemicals via surface water ingestion for any of the wildlife receptors is minor and negligible. Furthermore, ingestion of any water (via seep water exposure or TZW exposure) to wildlife receptor is expected to be a negligible for all wildlife receptors.

5.5.1.6 Study Area-Wide Exposure Scale

For hooded merganser, osprey, bald eagle, mink, and river otter, it was assumed that each wildlife receptor would forage throughout the Study Area (RM 2.0 to RM 11) and consume surface sediment and prey tissue from throughout the 9-mile stretch. EPA and its partners suggested that exposure areas of smaller scale should be evaluated for these wildlife receptors because foraging areas may be smaller than the Study Area. Specifically, EPA suggested that non-channel segments of the Study Area that were created for the Fate and Transport model (referred to as the fate and transport segments) should each represent an exposure area for wildlife. To consider whether smaller scales of exposure should be used for these wildlife receptors, it was first evaluated whether smaller scales of exposure would be ecologically relevant to the selected receptors, and then it was evaluated whether reducing the spatial scale of exposure would change HQ results and risk conclusions (regardless of ecological relevance).

The ecological relevance of reducing the exposure area was evaluated for each of the wildlife receptors:

- For hooded mergansers, no data were available on the foraging range of these receptors (Section 5.2.2.3) and therefore, it is unknown whether populations of mergansers are likely to forage within an area smaller than the Study Area. Hooded mergansers, unlike the other wildlife receptors evaluated using a Study Area-wide exposure area, primarily consume prey that could be limited to smaller exposure scales (i.e., clams and sculpin), and therefore, limiting exposure to smaller areas could be ecologically relevant for hooded mergansers.
- Bald eagles may forage along a shoreline stretch of that is less than the 9-mile Study Area (Section 5.2.2.4) and it is unknown whether osprey would forage within an area smaller than the Study Area (Section 5.2.2.5). The fish tissue used to calculate dietary exposure for bald eagle and osprey were all collected during Round 1 within the boundaries of the initial study area (ISA) between RM 3.5 and 9.2, so that the prey tissue for bald eagles and osprey actually represents a foraging distance along a six mile stretch. In addition, the fish species used primarily to model dietary prey consumption of bald eagles (i.e., carp and largescale sucker) and osprey (i.e., largescale sucker) cannot be confined to exposure areas smaller than the Study Area, because the home range of these fish species is greater than the Study area. Confining the prey tissue of bald eagles and osprey to areas smaller than the Study Area does not represent an ecologically relevant exposure area based on the home range of their prey.

- Mink have been reported to forage along small stretches of shoreline areas (ranging from 0.5 to 3.5 miles long; Section 5.2.2.6), and therefore, mink populations may likely forage in areas smaller than the Study Area. However, only a few of the prey items used to model the dietary exposure of mink can be confined to exposure areas smaller than the Study Area (i.e., sculpin, crayfish, each representing 15% of the diet). Therefore, confining the prey tissue of mink to areas smaller than the Study Area represents an ecologically relevant exposure area for mink foraging habitat and for mink prey that can be confined to small exposure areas; however, not all mink prey have a home range within an area smaller than the Study Area.
- River otter have been known to forage along shorelines on as small as six miles long (Section 5.2.2.7). The majority of the river otter diet (88%) was represented by fish collected during Round 1 within the boundaries of the initial study area (ISA) between RM 3.5 and 9.2, representing foraging distance along a six mile stretch. Therefore, the assumption that river otter forage throughout the Study Area where fish were collected is an ecologically relevant assumption.

Following the evaluation of the ecological relevance of reducing exposure areas to areas smaller than the Study Area, an uncertainty analysis was conducted to evaluate whether reducing the scale of exposure for sediment and prey consumption for hooded merganser, osprey, bald eagle, mink, and river otter would effect HQs. Reducing the scale of exposure for sediment ingestion by hooded merganser, osprey, bald eagle, mink, and river otter, to less than the Study Area does not significantly change HQs that were based on ingestion of sediment from throughout the Study Area. To evaluate this assumption, the maximum detected sediment concentration of Round 2 COPCs were used to calculate dietary dose HQs for all wildlife receptors, to determine an upper range estimate on what HQs could be based on sediments from smaller exposure areas. Using the single highest detected sediment concentration of the Study Area instead of the Study Area-wide sediment UCL results does not significantly change HQs for all wildlife receptors evaluated, except as noted for the following Round 2 COPC/receptor pairs:

- For hooded merganser, using maximum detected sediment concentrations from throughout the Study Area results in NOAEL HQs > 1.0 for lead, and benzo(a)pyrene. Also, the NOAEL HQ for dioxin TEQ more than doubled.
- For bald eagle, using maximum detected sediment concentrations from the Study Area results in NOAEL HQs > 1.0 for dioxin TEQ and mercury.

- For osprey, using maximum detected sediment concentrations from throughout the Study Area results in NOAEL HQs > 1.0 for dioxin TEQ, lead, and benzo(a)pyrene.
- For mink, using maximum detected sediment concentrations from throughout the Study Area results in NOAEL HQs > 1.0 for mercury and selenium, and also a LOAEL HQ > 1.0 for selenium. Also, the NOAEL and LOAEL HQs for dioxin TEQ more than doubled in magnitude for mink when the maximum sediment concentration was used.
- For river otter, using the maximum detected sediment concentration from throughout the Study Area causes the NOAEL HQ for dioxin TEQ to double in magnitude and also results in a LOAEL HQ > 1.0 for that chemical.

A similar uncertainty analysis was conducted to evaluate whether reducing the scale of exposure for prey consumption for hooded merganser, osprey, bald eagle, mink, and river otter would affect HQs. The maximum prey tissue concentrations were used to calculate dietary dose HQs for all wildlife receptors, to determine an upper range estimate on what HQs could be based on prey tissues from smaller exposure areas. Using the highest tissue concentrations of prey items within the Study Area instead of tissue EPCs based on Study Area-wide tissue UCLs does change the magnitude of some HQs, but does not change risk conclusions, except for the following Round 2 COPC/receptor pairs:

- For hooded merganser, using maximum prey tissue concentrations from throughout the Study Area results in a total PCB LOAEL HQ >1.0 and < 2.0 and a sum DDT LOAEL HQ > 1.0 and < 4.0. Additionally, the NOAEL HQ For sum DDTs more than doubled in magnitude with the use of the maximum prey tissue concentrations.
- For bald eagle, using maximum prey tissue concentrations from throughout the Study Area results in a mercury NOAEL HQ just over 1.0.
- For river otter, using prey tissue concentrations from throughout the Study Area results in a dioxin TEQ LOAEL HQ > 1.0 and < 2.0. However, because mink already have a LOAEL HQ > 1.0 for dioxin TEQ, this uncertainty with the dietary assumptions would not change the risk conclusions regarding dioxin TEQ risks for carnivorous mammals in the Study Area.

Based on this quantitative analysis, it was concluded that, even regardless of the ecological relevance of reducing exposure areas for hooded merganser, osprey, bald eagle, mink, and river otter, reducing prey tissue exposure scales to less than Study

Area-wide does not affect the HQ results for these wildlife receptors, except for several Round 2 COPC/receptor pairs, where minor changes in HQ results may result in additional Round 2 COPCs exceeding LOAEL TRVs for specific wildlife receptors.

5.5.1.7 Assumption of Surface Sediment from 20-ft Water Depth

All surface sediments from throughout the Study Area within the 20 ft (NAVD 88) water depth or shallower were used in the dietary dose calculations for hooded merganser, bald eagle, osprey, mink, and river otter. A depth of 0 to 20 ft was assumed to account for the lateral transport of sediments into the shallower habitats. To evaluate whether the use of a 0-to-5-ft sediment depth to represent the sediment exposure areas for hooded merganser, bald eagle, osprey, mink, and river otter could over or underestimate risks, dietary doses based on a UCL of sediment samples from within the 5-ft water depth were calculated.

NOAEL and LOAEL HQ exceedances were the same for all receptors evaluated based on the 0-to-20-ft depth sediment assumption as the 5-ft depth sediment assumption, except for the dioxin TEQ for mink. The use of the 0-to-5-ft sediment UCL decreased the HQ values by 75%, which caused the dioxin TEQ LOAEL HQs to be less than 1.0.

5.5.1.8 Ingestion Rate for Mink

A sediment ingestion rate of 9.4% was conservatively assumed for mink, based on the sediment ingestion rate measured for raccoons (Beyer et al. 1994). Due to the conservative nature of this sediment ingestion rate, the mink sediment ingestion rate of 9.4% was examined to determine if a lower incidental sediment ingestion rate would result in changes in HQ exceedances.

Of the three chemicals with HQs exceeding 1.0 for mink (PCB TEQ, total PCBs, and dioxin TEQ), no changes occurred for PCB TEQ or total PCBs since the majority of the risk associated with these analytes can be attributed to prey tissue consumption (see Table 5-62). For dioxin TEQ, over 75% of the risk associated with dioxin TEQ can be attributed to sediment ingestion (see Table 5-62), and a decreased sediment ingestion rate reduced HQs. However, the incidental sediment ingestion rate of mink would need to be less than 0.75 % in order for the dioxin TEQ LOAEL HQ to be less than 1.0. Therefore, dioxin TEQ risks to mink are expected, even if the sediment ingestion rate assumption is decreased to <1%.

5.5.1.9 100% Site Use

A site use factor of 1.0 was assumed for all wildlife receptors, which may overestimate site use by migrating birds (i.e., osprey) or birds that likely forage outside the Study Area (i.e., hooded merganser and bald eagle). Because hooded mergansers and bald eagles may forage in nearby aquatic and terrestrial environments in addition to the Study Area, and because during the migratory season, non-breeding osprey forage outside the Study Area, a site use factor of 0.5 was also used to estimate exposure for each of these receptors to provide a range of HQs. A

comparison of HQs based on site use factors of 1.0 and 0.5 for hooded merganser, bald eagle, and osprey based on the dietary dose approach are presented in Table 5-68.

For the dietary dose approach, assuming a site use factor of 0.5 would reduce the number of Round 2 COPCs exceeding NOAELs. For hooded merganser, the mercury, total PCB, sum DDE, and total DDT NOAEL HQs would be ≤ 1.0 in the dietary approach assuming a site use factor of 0.5. For bald eagle, the total PCB NOAEL HQs would be < 1.0 in the dietary approach assuming a site use factor of 0.5.

5.5.1.10 Use of UCL to Represent Tissue and Sediment EPCs

In dietary dose calculations, UCLs of prey tissue and sediment concentrations were used to represent EPCs. The UCL is a conservative estimate of the true population mean, and differences between the sample population mean and UCL decrease with data homogeneity and sample size. To evaluate whether the sample means for prey tissue and sediment concentrations result in substantially different risk calculations, dietary dose estimates were calculated using mean concentrations and HQs based on UCL EPCs were compared to HQs based on mean EPCs[52]. The change in HQs was greater than 50% for most chemicals, except for metals where the use of mean values only slightly reduced HQ values. Tables 5-69, 5-70, and 5-71 show the comparison of HQs based on mean EPCs to HQs based on UCL EPCs for spotted sandpiper, Study Area-wide foraging birds (hooded merganser, bald eagle, and osprey), and carnivorous mammals (mink and river otter), respectively.

For sandpiper, HQs calculated using the mean EPCs and UCL EPCs were compared for both the clam and worm dietary scenarios (Table 5-69). For the clam diet, HQs based on mean EPCs also resulted in NOAEL HQs > 1.0 for two Round 2 COPCs: chromium and PCB TEQ. For the worm diet, HQs based on mean EPCs also resulted in NOAEL HQs > 1.0 for four Round 2 COPCs: total PCBs. NOAEL and LOAEL HQs were < 1.0 for all other Round 2 COPCs based on the worm and clam diets.

HQs based on mean EPCs also results in NOAEL HQs > 1.0 for hooded merganser/mercury, hooded merganser/sum DDT, and osprey/mercury, and HQs, < 1.0 for all other Round 2 COPCs for these two receptors (Table 5-70). HQs based on mean EPCs results in NOAEL and LOAEL HQs < 1.0 for all bald eagle Round 2 COPCs.

For mammals, HQ values decreased by over 40% when using EPCs based on mean concentrations compared to HQs based on UCL EPCs (Table 5-71). With the use of mean EPCs, LOAEL HQ exceedances were < 1.0 for PCB TEQ and for dioxin TEQ; however, LOAEL HQs were > 1.0 for both mink and river otter based both UCL EPCs and mean EPCs.

[52] Only HQs based on UCL EPCs greater than one were compared to HQs based on mean EPCs. HQs < 1.0 based on UCL EPCs would also have HQs < 1.0 based on mean EPCs.

5.5.1.11 Shorebird Exposure Scale

Risks to shorebirds were evaluated for each shorebird habitat area based on the assumption that shorebird populations would breed in these habitat areas and forage on sediments and benthic tissue only from within these areas. This assumption is highly conservative and assumes that the quality of habitat at each beach is high enough for breeding. Shorebird beaches were evaluated for quality of foraging habitat; however, shorebird beaches were not evaluated for quality of breeding habitat.

The assumption that shorebirds would breed within each of the habitat areas and would only consume sediments and prey tissue from within each habitat areas (and not in adjacent beached) likely overpredicts actual risks to shorebird populations in the Study Area. Further information regarding the breeding habitat potential of each of the shorebird beach areas could help identify which beaches represent realistic risk estimates for shorebirds (i.e., for which beaches is the assumption that shorebirds would breed there is realistic).

5.5.1.12 Benthic Prey Tissue Data

There is uncertainty associated with the use of only laboratory-exposed worm tissue to represent the shorebird diet. Dietary dose estimates are higher based on laboratory-exposed worms than those based on field-collected clams and may overestimate risks to shorebirds. Using laboratory-exposed worm tissue to represent the prey, six Round 2 COPCs (i.e., PCB TEQ, total PCBs, dioxin TEQ, aldrin, and sum DDD) had LOAEL HQs greater than 1.0 at individual beaches; whereas, using field-collected clam tissue to represent prey, only one Round 2 COPC (i.e., total PCBs) had LOAEL HQs greater than 1.0 at one beach location.

There is also uncertainty associated with the use of the FWM and site-specific regressions to estimate benthic tissue at the five beach locations (B1, B4, B7, B8, and B14) with no tissue data collected. The uncertainties associated with theses methods are presented in Appendix E. The predicted tissue Round 2 COPC concentrations based on FWM or site-specific regressions may over or underpredict toxicity to shorebirds.

A UCL tissue concentration was used to estimate benthic tissue at these five beach locations (B1, B4, B7, B8, and B14) where the FWM could not be used or where a site-specific regression could not be developed. A UCL was used to represent clam and/or worm tissue for 13 Round 2 COPCs: arsenic, cadmium, chromium, copper, lead, mercury, selenium, total PAHs, endrin, endrin aldehyde, endrin ketone, BEHP, and dibutyl phthalate. The use of a Study Area-wide UCL for benthic prey tissue may over or underpredict benthic tissue concentrations at individual beach locations.

5.5.2. Bird Egg Assessment

There are uncertainties associated with the effects and exposure assessment of the bird egg LOE. The primary uncertainties associated with the effects assessment of

bird egg approach are the selected TRVs and the TEQ approach for dioxins and dioxin-like PCB congeners. There is additional uncertainty associated with the BMF and fish tissue assumptions used to estimate bird egg tissue concentrations and the assumption that site use factors are equal to 1.0 for both piscivorous bird receptors (i.e., bald eagle and osprey). Each of these uncertainties is discussed and evaluated in the following subsections.

5.5.2.1 Selected TRVs

All bird egg Round 2 COPCs have TRVs based on field studies, in which other chemicals are known to be or are likely to be present and likely to contribute to observed toxicity. Field-based egg TRVs may represent toxicity thresholds that overestimate risks to birds; however, more toxicological data for bird eggs are based on field data, and these field data provide toxicological data on relevant bird species that are present in the LWR. The uncertainties of the selected bird egg TRVs for each Round 2 COPC are summarized in Table 5-73 through 5-78.

5.5.2.2 TEQ Approach

The uncertainties with the TEQ approach are the same as those described for the dietary approach (Section 5.5.2.1.3).

5.5.2.3 Modeled Bird Egg Tissues

In the bird egg approach, concentrations of Round 2 COPCs were estimated in egg tissues using a literature-derived BMF and assuming 100% ingestion of fish prey. BMFs were based on Willamette-specific data, when available; however, a great deal of variability was reported in BMFs reported from various locations. All BMFs were based on osprey data from the Willamette River except for the mercury BMF, which was based on bald eagle date from the Lower Columbia River. No BMFs were available from the Willamette River for bald eagles. The Willamette River-specific BMFs used for PCB TEQ, dioxin TEQ, total PCBs, and sum DDE are based on osprey that are present on the Willamette River during breeding season but may overwinter in Central America, where they may be exposed to chemicals. Therefore, the BMFs may overpredict osprey egg tissue concentrations from chemical exposure within the Study Area only.

There is also uncertainty with the assumption that bald eagles consume only fish. Bald eagles are known to consume fish as well as other birds and occasionally small mammals. Watson et al. (1991) reports the diet composition of bald eagles in the Columbia River Estuary as 90% fish, 7% avian, and 3% mammalian. Birds and mammals used as prey by the bald eagle may contain higher concentrations of bioaccumulative Round 2 COPCs in their tissue than fish if the birds and mammal consumed are consumers of fish. The assumption of 100% ingestion of fish by bald eagle may over or underestimate risk based on modeled bird egg tissues.

In order to address this uncertainty, HQs were recalculated assuming a bald eagle diet of 90% fish and 10% birds. Calculations of dioxin TEQ, PCB TEQ, total PCBs, DDE, DDT, and mercury concentrations in bird prey tissue were made using UCL fish prey

tissue concentrations and bird tissue: fish tissue BMF factors reported in the literature. BMFs for total PCBs, sum DDE, sum DDT, dioxin TEQ, and PCB TEQ were based Braun and Norstrom (1989) where concentrations in whole body alewife prey tissue and whole body herring gull tissue were measured. A BMF of 9.2 for mercury based on a study of osprey nestlings fed prey with known mercury concentrations was reported in DesGranges et al. (1998).

When dose calculations were made for dioxin TEQ, PCB TEQ, total PCBs, sum DDE, sum DDT, and mercury using a bald eagle diet consisting of 90% fish and 10% birds, overall risk conclusions for total PCBs was impacted most significantly with both the LOAEL and NOAEL HQs greater than 1.0. Therefore, the total PCB HQs for bald eagles based on the consumption of fish prey only may underestimate risks, and total PCBs may pose a risk to bald eagles. The NOAEL HQs for PCB TEQ, mercury, and DDE were all greater than one and less than four based on the new dose calculation; however, LOAEL HQs for these Round 2 COPCs were less than one, and it is likely that risk conclusions for bald eagles would not change for these Round 2 COPCs. There was no change to risk conclusions (all HQs were < 1.0) for total DDTs and dioxin TEQ based on the dose calculation derived from a mixed fish and bird diet.

5.5.2.4 100% Site Use

A site use factor of 1.0 was assumed, which may overestimate site use by migrating birds (i.e., osprey) or birds that likely forage outside the Study Area (i.e., bald eagle). Because bald eagles may forage in nearby aquatic and terrestrial environments in addition to the Study Area and because during the migratory season, non-breeding osprey forage outside the Study Area, a site use factor of 0.5 was also used to estimate exposure for each of these receptors to provide a range of HQs. A comparison of HQs based on site use factors of 1.0 and 0.5 for hooded merganser, bald eagle, and osprey using the bird egg approach are presented in Table 5-72. Assuming a site use factor of 0.5 would reduce the magnitude of NOAEL and LOAEL HQs for bald eagle and osprey; however, risk conclusions would remain the same as with a site use factor of 1.0.

5.5.3 Summary of Uncertainties

The associated uncertainties for all Round 2 COPCs are presented in Tables 5-73 through 5-78 for each receptor evaluated. These uncertainties were used along with the results of the HQ calculations (Section 5.4.1) to make risk conclusions and identify Round 2 iCOCs for wildlife receptors.

5.6 RISK CONCLUSIONS

The section presents the risk conclusions made for wildlife receptors and the Round 2 iCOCs identified for wildlife. The results of the HQs presented in Section 5.3.1 were combined with the uncertainties associated with each Round 2 COPC/receptor presented in Section 5.3.2 to arrive at risk conclusions. The selected iCOCs have been

identified for this Round 2 ERA and final COCs will be identified as part of the BERA.

Round 2 iCOCs were identified for wildlife as those Round 2 COPCs that exceeded LOAEL TRVs for at least one Round 2 COPC/receptor risk scenario, or where NOAEL TRVs were exceeded for those receptors selected to be representative of special-status species (i.e., bald eagles). Based on this rationale, the following chemicals were identified as iCOCs for wildlife:

PCBs – Total PCBs and PCB TEQ were identified as iCOCs (LOAEL HQs > 1.0) for spotted sandpiper, mink, and river otter based on the dietary dose LOE and for bald eagle and osprey (NOAEL and LOAEL HQs > 1.0) based on the bird egg LOE. Total PCB and PCB TEQ risks to shorebirds can be isolated to six localized beach habitat areas with high uncertainty (assuming 100% use of beach habitat area and 100% ingestion of laboratory-exposed worms[53]). No risks were estimated for osprey and bald eagle based on the dietary dose LOE.

Dioxins and furans – Dioxin TEQ was identified as an iCOC (LOAEL HQ > 1.0) for spotted sandpiper and mink based on the dietary dose LOE and for bald eagle and osprey (NOAEL and LOAEL HQs > 1.0) based on the bird egg LOE. Dioxin TEQ risks to shorebirds can be isolated to two localized beach habitat areas with high uncertainty (assumes 100% use of beach habitat area and 100% ingestion of laboratory-exposed worms). No risks were estimated for shorebirds based on ingestion of clams or for osprey and bald eagle using the dietary dose LOE.

Sum DDE – Sum DDE was identified as an iCOC (NOAEL and LOAEL HQs > 1.0) for piscivorous birds based on the bird egg approach for osprey and bald eagle. Using the dietary dose LOE, no risks to osprey or bald eagle were estimated.

Sum DDD – Sum DDD was identified as an iCOC for shorebirds at two localized beach areas (where LOAEL HQs = 1.6 and 1.2) with high uncertainty because risks were determined assuming 100% use of beach habitat area and 100% ingestion of laboratory-exposed worms and because of the low magnitude of HQ exceedance. No risks were estimated for shorebirds based on ingestion of clams.

Aldrin – Aldrin was identified as an iCOC for shorebirds at one localized beach area (where LOAEL HQ = 1.1) with high uncertainty because risks were determined assuming 100% use of beach habitat area and 100% ingestion of laboratory-exposed worms (as represented by bioaccumulative tissue), and because of the low magnitude of HQ exceedance. Furthermore, no risks were estimated for shorebirds based on ingestion of clams.

[53] When the shorebird diet was modeled using 100% ingestion of field-collected clams, risks to shorebirds were identified at only one shorebird beach area (where the LOAEL HQ > 1.0).

All other Round 2 COPCs were not identified as iCOCs or recommended to be carried forward to the BERA because risks to wildlife are not expected.

Wildlife iCOCs identified were mapped (presented in Section 10 of the Comprehensive Round 2 Report) to identify initial areas of potential concern (AOPCs) for wildlife receptors. Potential data needs for wildlife in the BERA are presented in Section 8.0.

6.0 AMPHIBIAN AND REPTILE RISK ASSESSMENT

This section presents the Round 2 ERA for amphibian and reptile receptors in the Portland Harbor Study Area. Section 6.1 identifies the receptors of concern, exposure pathways, and LOEs evaluated for determining potential risks to amphibian and reptile populations. The results, including the identification of amphibian and reptile Round 2 COPCs, and conclusions of the screening assessment are presented in Section 6.2. The conclusions of the screening assessment for amphibians and reptiles will inform the BERA.

6.1 PROBLEM FORMULATION

This section summarizes the problem formulation for amphibians and reptiles in the Study Area. A detailed discussion of the problem formulation is presented as Appendix B of the Programmatic Work Plan (Integral et al. 2004b).

6.1.1 Amphibian and Reptile Resources Potentially at Risk

There is a paucity of scientific information on the occurrence of amphibians and reptiles in the LWR. Table 6-1 lists the amphibians and reptiles that could be present in or near the Study Area. Appendix B of the Programmatic Work Plan (Integral et al. 2004b) provides additional information on the prey and habitat preferences of amphibians and reptiles, as well as their potential use of the LWR. Of the species listed in Table 6-1, one amphibian species (northern red-legged frog) and two reptile species (painted turtle and western pond turtle) are special-status species.[54]

Conditions within the LWR are likely to provide a limited amount of habitat suitable for amphibians and reptiles listed. Most of the native amphibians (e.g., long-toed salamander, northern red-legged frog and Pacific tree frog) that may be found in the Study Area prefer undisturbed areas that offer ephemeral wetlands with emergent vegetation and shallow waters. They are preyed upon by the more ubiquitous bullfrogs, which are invasive to the Pacific Northwest and have few predators.

In the LWR, painted turtles may typically be found in sloughs and ponds which provide shallow aquatic areas with open banks and abundant plant growth. The most frequently encountered reptiles on the LWR valley floor are the common garter and northwestern garter snakes. Both species prefer wet vegetated terrestrial habitats, where they may be found lying under rocks, wood and grasses. Roadside ditches or embankments may also provide a suitable place for either species to reside.

An amphibian/reptile reconnaissance survey was conducted in the summer of 2002 to confirm the presence of amphibians within the Study Area (Table 6-2). During the survey, only the northern red-legged frog was observed (Integral et al. 2004b). As a result of the survey, potential amphibian habitat areas were identified on the LWR as

[54] Special-status species include federal and state proposed and candidate species.

low-sloping beaches and steeper, riprapped or rocky banks as presented in Appendix B of the Programmatic Work Plan (Integral et al. 2004b). These amphibian habitat areas were superimposed over the locations of quiescent areas within the Study Area to identify the sampling areas relevant for amphibian exposure in the Round 2 surface water field sampling plan (FSP) (Integral 2004). However, not all of the quiescent areas were selected as a surface water sampling locations. Representative quiescent areas and the amphibian surface water sampling locations are shown on Figure 6-1.

Although terrestrial habitat requirements for reptiles may be available near the Study Area, reptiles were not observed during the 2002 survey; and it was concluded that the reptile exposure pathway is incomplete due to the lack of suitable habitat within the Study Area. Therefore, risks to reptiles were not evaluated and as stated in Appendix B of the Programmatic Work Plan (Integral et al. 2004b). Amphibian risks were evaluated at the population level, as stated in Appendix B of the Programmatic Work Plan because no threatened or endangered species are present within the Study Area.

6.1.2 Receptors of Concern and Exposure Pathways

Potential chemical exposure pathways for amphibian receptors within the Study Area were identified in Appendix B of the Programmatic Work Plan (Integral et al. 2004b) and direct contact with or ingestion of sediments or surface water or indirect exposure through the ingestion of prey (i.e., biota). The preliminary ecological CSM for amphibians and reptiles (Figure 6-2) that was developed for the Programmatic Work Plan illustrates the possible exposure pathways that chemicals may follow from potential sources to receptors.

Direct contact with surface water is a complete and major pathway for amphibians and was the pathway considered in this risk assessment. Surface water ingestion is a complete but relatively minor pathway for amphibians. TZW (porewater) direct contact and ingestion pathways are incomplete and were not evaluated further. The biota ingestion pathway is considered a complete and major pathway for amphibians because some species prey on benthic invertebrates; however, no toxicological data for amphibians and ingestion pathways were available, so these pathways were not evaluated. Direct contact with sediment varies depending on the species and life stage. It is a complete and uncertain pathway for amphibians, and insufficient data were available to quantitatively evaluate this pathway.

6.1.3 Lines of Evidence Approach and Methods

LOEs for assessing risks to amphibians were presented in Appendix B of the Programmatic Work Plan (Integral et al. 2004b) and further refined based on discussions between LWG, EPA, and EPA's partners. LWG agreed to evaluate the surface water exposure pathway (Table 6-3), by comparing surface water

concentrations from relevant exposure areas (Figure 6-1) to literature-based values or AWQC.

6.2 SUMMARY OF SCREENING ASSESSMENT

Surface water data were screened for the identification of Round 2 COPCs for amphibians by comparing maximum detected chemical concentrations in surface water in relevant amphibian exposure areas (Figure 6-1) to Eco SLs. These Round 2 COPCs for amphibians were further evaluated in the screening assessment. The results of the amphibian screening assessment of surface water are presented in Attachment G7.

Six COIs were identified as Round 2 COPCs for amphibians: zinc (dissolved), 4-chloro-3-methylphenol, total PCBs, 2,4′-DDT, 4,4′-DDT, and total DDTs; however, none of these six Round 2 COPCs were identified as amphibian iCOCs for the following reasons:

- Amphibian-specific toxicity data for total PCBs and total DDTs were several orders of magnitude higher than the Eco SLs and are more relevant to evaluate the potential risks to amphibians. None of these chemicals were detected at concentrations greater than the amphibian-specific toxicity criteria.
- Exceedances of the chronic Eco SL was only identified at one location within the Study Area for total PCBs, zinc, 2,4′-DDT, 4,4′-DDT, and 4-chloro-3-methylphenol. Thus, based on the spatial extent of exceedances, these chemicals would not pose a risk to amphibians at the population level.

No iCOCs were identified for amphibians and reptiles. This receptor group was not further evaluated and no potential data needs were identified. This receptor group will be summarized in the BERA; however, no further analysis will be conducted. Any additional surface water collected prior to the baseline risk assessment will be incorporated into an updated screening analysis for amphibians, to better understand any influence of flow conditions on surface water concentrations.

7.0 AQUATIC PLANTS RISK ASSESSMENT

This section presents the Round 2 ERA for aquatic plant receptors in the Portland Harbor Study Area. Section 7.1 identifies the receptors of concern, exposure pathways, and LOEs evaluated for determining risks to aquatic plant species present in the LWR. The results, including the identification of amphibian and reptile Round 2 COPCs, and conclusions of the screening assessment are presented in Section 7.2. The conclusions of the screening assessment for aquatic plants will inform the BERA.

7.1 PROBLEM FORMULATION

This section presents the ecological receptors, exposure pathways, and LOEs evaluated for determining risks to aquatic plant species present in the LWR.

7.1.1 Aquatic Plants Resources Potentially at Risk

Current conditions within the LWR prevent the successful establishment of dense, submerged and emergent aquatic plant communities along the river banks. Factors such as high turbidity and wave action, the development and stabilization of the banks and shoreline, and the establishment of exotic plant species likely prevent the growth of healthy native aquatic plant communities in the LWR. To date, no comprehensive vegetation surveys have been conducted within the Study Area to quantify and describe the plant communities. However, two qualitative plant community surveys have been conducted in the LWR (Adolfson et al. 2000; Integral et al. 2004b), and a list of species that were observed during those surveys is presented in Table 7-1.

Adolfson et al. (2000) conducted an inventory of fish and wildlife habitats along the shoreline of the LWR and identified 10 distinct habitat classes: bottomland forest, foothill savanna, conifer forest, meadow, shrub, emergent wetland, beach, rock outcrop, open water, and unvegetated/disturbed. Although all of these habitats are present in the vicinity of the LWR, the bottomland forest, emergent wetlands, beach, and open water habitat classes are the most common, occurring along the shoreline within the Study Area. Historically, bottomland forests were an important component of the Willamette River floodplain system (Sedell and Froggatt 1984), but they have been reduced to a smaller percentage of their former extent (Adolfson et al. 2000). Emergent wetlands also exist in a few small remnant patches in areas adjacent to the shoreline. Beach habitats throughout the LWR typically consist of narrow shoreline areas with sand substrate and are dominated by a variety of annual grasses and perennial shrubs. Open-water habitats exist throughout the LWR in tributaries, sloughs, and side channels and are often dominated by aquatic plant species from bottomland forest, emergent wetland, and scrub/shrub plant communities (Adolfson et al. 2000).

Potential aquatic plant habitats were characterized during the aquatic plant and amphibian/reptile reconnaissance survey (Integral et al. 2004b) and included the identification of submerged and emergent aquatic plant species throughout the Study

Area. Twenty-six plant species were identified at the Study Area during this survey, most of which were obligate and facultative wetland plant species. Half of the plant species identified are exotic to the LWR (Table 7-1). Nine special-status plant species are present in the Willamette Valley and may be present in the LWR, though they were not observed during the reconnaissance survey.

7.1.2 Receptors of Concern and Exposure Pathways

Aquatic plants observed within the Study Area included submergent plants, emergent herbaceous and woody plants, shrubs, and trees along the shoreline of the LWR, particularly in quiescent areas identified as amphibian and aquatic plant habitat in Figure 7-1. Exposure of aquatic plants was assessed at the community level and includes all types of aquatic plants present in the LWR.

Potential chemical exposure pathways for aquatic plants within the Study Area were identified in Appendix B of the Programmatic Work Plan (Integral et al. 2004b) and further refined based on discussions between LWG, EPA, and EPA's partners (EPA 2004a, 2005a). The preliminary ecological CSM for aquatic plants that was developed for the Programmatic Work Plan illustrates the possible exposure pathways that chemicals may follow from potential sources to receptors (Figure 7-2). The exposure pathways identified for the aquatic plants include direct contact with sediment, surface water, and TZW by means of root uptake.

Because aquatic plants actively and passively transfer chemicals from surface water and sediments, both of these pathways (i.e., direct contact with surface water and sediment) are considered complete and major pathways of exposure for the aquatic plant communities within the Study Area (Figure 7-2). Because TZW is a combination of groundwater and porewater, both rooted and floating aquatic plants may have direct contact and be exposed to chemicals via this pathway as well; however, the extent of that contact is uncertain and spatially variable. Therefore, the exposure of aquatic plants to chemicals in TZW is considered an incomplete pathway because the importance of this pathway to the aquatic plant community is uncertain (Figure 7-2).

7.1.3 Lines of Evidence Approach and Methods

LOEs for assessing risks to aquatic plants were presented in Appendix B of the Programmatic Work Plan (Integral et al. 2004b) and further refined based on discussions between LWG, EPA, and EPA's partners. LWG agreed to evaluate the surface water exposure pathway (Table 7-2), by comparing surface water concentrations from relevant exposure areas (Figure 7-1) to literature-based values or AWQC. Exposure areas for aquatic plants are defined as quiescent, backwater areas within the Study Area and are the same as those defined as amphibian habitat (Section 6.1.1). The aquatic plant exposure areas are shown on Figure 7-1, along with the 14 surface water sampling locations established for the Round 2 surface water and field sampling FSP (Integral 2004). The TZW pathway was evaluated per the request

of EPA and assessed in the same manner as surface water by comparing chemical concentrations in TZW to Eco SLs.

LOEs for assessing risks to aquatic plants were presented in Appendix B of the Programmatic Work Plan (Integral et al. 2004b) and further refined based on discussions between LWG, EPA, and EPA's partners. LWG agreed to evaluate the surface water exposure pathway (Table 7-2), by comparing surface water concentrations from relevant exposure areas (Figure 7-1) to literature-based values or AWQC. Exposure areas for aquatic plants are defined as quiescent, backwater areas within the Study Area and are the same as those defined as amphibian habitat (Section 6). The aquatic plant exposure areas are shown on Figure 7-1, along with the 14 surface water sampling locations established for the Round 2 surface water and field sampling FSP (Integral 2004). The TZW pathway was evaluated per the request of EPA and assessed in the same manner as surface water by comparing chemical concentrations in TZW to Eco SLs.

A comparison of bulk sediment concentrations to relevant TRVs representing plant toxicity was proposed to assess aquatic plant exposure to chemicals in sediments (Table 7-2). In an evaluation of plant toxicity from exposure to chemicals in surface water, porewater, and sediments in the Calcasieu Estuary, it was determined that surface water and porewater were more important pathways of exposure to aquatic plants than sediment and therefore, the sediment pathway was not evaluated in the Calcasieu Estuary BERA (MacDonald ES 2002, Appendix D). A literature search for studies examining the exposure of aquatic plants to chemicals via sediment produced no relevant toxicity data. As a result, this LOE was not evaluated because no relevant toxicity data were available for the evaluation of the exposure of aquatic plants to chemicals in sediment.

7.2 SUMMARY OF SCREENING ASSESSMENT

Surface water and TZW data were screened for the identification of Round 2 COPCs for aquatic plants by comparing maximum detected chemical concentrations in surface water in relevant aquatic plant exposure areas and in TZW at nine locations to Eco SLs (Figure 7-1). TZW Round 2 COPCs were further evaluated to determine potential iCOCs. These Round 2 COPCs in surface water and potential iCOCs in TZW were further evaluated in the screening assessment. The results of the aquatic plant screening assessment of surface water and TZW are presented in Attachment G8.

Six surface water Round 2 COPCs were identified as Round 2 COPCs for aquatic plants: zinc (dissolved), 4-chloro-3-methylphenol, total PCBs, 2,4′-DDT, 4,4′-DDT, and total DDTs. Twenty-four potential iCOCs in TZW also were identified for aquatic plants, including PAHs (i.e., 2-methylnaphthalene, acenaphthene, anthracene, benzo[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene, benzo[g,h,i]perylene, benzo[k]fluoranthene, chrysene, dibenzo[a,h]anthracene, fluoranthene, fluorene,

indeno[1,2,3-cd]pyrene, naphthalene, phenanthrene, pyrene), DDTs (i.e., 2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDE, 4,4′-DDT, and total DDTs), cyanide, and perchlorate. However, none of these chemicals were identified as aquatic plant iCOCs for the following reasons:

- Locations of TZW samples are often not located within quiescent habitat areas identified for aquatic plants and may be in deeper waters where rooted aquatic plants are not likely to grow and be exposed. Furthermore, because localized exceedances of Eco SLs are limited within the quiescent areas, these chemicals would not pose a risk to aquatic plants at the community level.
- Exceedances of the chronic Eco SLs for total PCBs, zinc, 2,4′-DDT, 4,4′-DDT, and 4-chloro-3-methylphenol in surface water were observed at only one location in the Study Area. Thus, these chemicals would not pose a risk to aquatic plants at the community level.
- Plant-specific toxicity data for the majority of the Round 2 COPCs and potential iCOCs identified for surface water and TZW (including total PCBs, total DDTs, acenaphthene, fluoranthene, naphthalene, phenanthrene, cyanide, and perchlorate) were greater than the Eco SLs, which suggests that risks based on the Eco SLs may be overestimated for aquatic plants.
- The aquatic plant community of the LWR consists of species that are expected to exist in the habitat of an industrial harbor providing additional evidence that risks to aquatic plants at the Study Area are not significant at the community level.

No iCOCs were identified for aquatic plants. This receptor group was not further evaluated and no potential data needs were identified. This receptor group will be summarized in the BERA; however, no further analysis will be conducted. Any additional surface water collected prior to the baseline risk assessment will be incorporated into an updated screening analysis for aquatic plants, to better understand any influence of flow conditions on surface water concentrations.

8.0 RISK CHARACTERIZATION SUMMARY AND DATA GAPS ANALYSIS

This section summarizes the risk characterization, including iCOCs for each receptor group and identifies potential data needs. Data needs represent information or data required to complete the BERA. Filling those information needs will address key uncertainties that affect the outcome of the risk evaluation and the spatial scale over which risks need to be assessed to support the selection of a remedy for the Portland Harbor Superfund Site. Key uncertainties include those related to the data and assumptions used to assess risks.

Data needs for the ERA are grouped into the following categories:

- Analytical data that can be filled by the collection and analysis of additional samples.
- Further analysis of existing data (e.g., development of predictive modeling).
- Additional review of existing literature, including regional data, toxicological data, and other information to refine exposure estimates, toxicological thresholds, and ultimately, risk estimates.

Broader uncertainties associated with the state of the science or environmental policy and guidance are not included. There is a degree of uncertainty in any quantitative risk assessment. The exposure and toxicity assumptions used for this risk assessment, which were based on EPA guidance, current scientific literature, and best scientific judgment, are inherently uncertain. Therefore, the resulting risk estimates carry a degree of uncertainty.

iCOCs were identified through the Round 2 ERA for each receptor group and are presented in the following subsections, by receptor group. The identification of iCOCs used less-conservative assumptions than those used in the SLERA and incorporated a weighted approach across LOEs in an effort to narrow the list of chemicals and identify the potential areas associated with risks to ecological receptors in the Study Area. Final COCs will be identified as part of the BERA. For all Round 2 ERA iCOCs, an initial preliminary sediment goal (iPRG) was developed, except for those iCOCs where no relationship between sediment and tissue concentrations could be developed. For those iCOC-receptor pairs for which an iPRG could be developed, areas within an ecologically appropriate home range specific to each receptor (i.e., area-specific or site-wide) were identified that were determined to potentially pose risk. These areas were called initial areas of potential concern (iAOPCs) and are presented in Section 10 of the Comprehensive Round 2 Report.

8.1 BENTHIC INVERTEBRATES

The following section presents the conclusions of the benthic invertebrate assessment as determined by this Round ERA.

8.1.1 Summary of Round 2 Risk Conclusions

Multiple LOEs were used to assess risks to benthic invertebrates (Section 3.0), including the following: 1) sediment toxicity testing and modeling (FPM and LRM, 2) critical tissue-residue assessment, 3) surface water assessment, and 4) TZW assessment, including measured TZW concentrations and predicted TZW concentrations based on the EqP approach. In the screening assessment of benthic invertebrates, one additional LOEs was evaluated; sediment chemistry data were compared to literature-based SQVs (both consensus-based and empirical SQVs).

The FPM in combination with toxicity testing was identified as providing the best prediction of areas posing risks to the benthic community. Because the FPM cannot establish causality between chemicals and toxicity, the chemicals associated with toxicity in the FPM were identified as potential iCOCs in this Round 2 ERA. The results of the toxicity testing and predictions of toxicity based on the FPM are presented in Figure 3-8. Potential iCOCs for benthic invertebrates based on the FPM were the same as the Round 2 COPCs identified by the FPM as those chemicals statistically associated with toxicity to the benthic community, including: arsenic, cadmium, lead, mercury, silver, zinc, dibutyl phthalate, PCBs, DDTs (as sum DDD, sum DDE, and sum DDT), dieldrin, total chlordane, alpha-HCH, beta-HCH, delta-HCH, endrin ketone, diesel-range hydrocarbons, residual-range hydrocarbons, ammonia, and sulfides. In addition, percent fines was identified by the FPM as a physical condition that could affect the benthic community structure. The use of diesel-range hydrocarbons, residual-range hydrocarbons in the FPM is uncertain and will be re-evaluated in the BERA.

Several uncertainties are associated with the risk characterization of the benthic community based on the FPM, some are related to the structure of the model, and others are related to the data used for making predictions based on the model. Uncertainties associated with model structure include the mathematical assumption that the concentration of each chemical is independent of the concentrations of other chemicals and the summing methods of chemicals. Uncertainties associated with the data used in the model include lack of causality due to a mixture of chemicals in the sediment samples, selection of toxicity tests and endpoints used in the model, and limitations in suite of chemicals analyzed. The effect of these uncertainties on the risk estimate is unknown.

Benthic invertebrate iCOCs were identified based on the tissue, surface water, and TZW LOEs and included the following: cadmium, copper, zinc, total PAHs, three individual PAHs (benzo(a)pyrene, chrysene, and pyrene), total PCBs, and DDTs (including 2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDE, 4,4′-DDT, and total DDTs).

Cyanide, perchlorate, 13 individual PAHs,[55] TBT, and BEHP were also retained as potential iCOCs for benthic invertebrates. Potential iCOCs for benthic invertebrates were those chemicals that were associated with high uncertainty or identified by the TZW LOE, but not supported by other LOEs, as posing risks to benthic invertebrates. These potential iCOCs are not expected to pose significant risks to benthic invertebrates because there is limited potential for exposure.

The selected TRVs and Eco SLs and the relevance of the TZW pathway to benthic invertebrates are the primary uncertainties associated with the tissue, surface water, and TZW risk characterization for the benthic community. Because the majority of the TRVs and Eco SLs were conservative screening values, the risk estimates based on these values are likely overestimated. There is uncertainty associated with the TZW assessment based on the selection of TZW sampling sites at potential source areas and whether TZW data represent appropriate ecological exposure for the benthic community, which may result in an overestimate of risk.

The areas identified as potentially causing toxicity using either direct benthic toxicity or the FPM, will be evaluated in more detail in the BERA. Further analysis (i.e., cluster analysis) of the direct benthic toxicity results and FPM is presented in Section 10 of the Comprehensive Round 2 Report to identify iAOPCs for benthic invertebrates. The majority of areas identified by the FPM as potential causing adverse effects to the benthic community were also identified through the tissue and water LOEs. Areas identified as "indeterminate" or with elevated tissue residues, will also be evaluated in more detail in the BERA. In those cases where benthic invertebrate laboratory toxicity measurements or FPM were not definitive, the potential for adverse effects as indicated by tissue concentrations in excess of toxicity reference values (tissue residue) or water (either near-bottom or TZW) concentrations in excess of water quality criteria or effects-based screening levels will be used to resolve benthic risks.

8.1.2 Potential Data Needs

The primary sampling need identified for the benthic assessment is potentially to conduct additional toxicity testing in areas identified as indeterminate. The predicted toxicity LOE identified areas as indeterminate based on FPM (exceeding Level 2 but less than Level 3). Other LOEs were used to assess risks to benthic invertebrates in these areas; however, additional toxicity testing may be warranted in areas where the additional LOEs did not alter the indeterminate status. This analysis will be determined and conducted, if warranted, for analysis in the BERA. No additional sampling needs were identified for other LOEs.

[55] The following PAHs were identified as iCOCs for benthic invertebrates: 2-methylnaphthalene, acenaphthene, anthracene, benzo[a]anthracene, benzo[b]fluoranthene, benzo[g,h,i]perylene, benzo[k]fluoranthene, dibenzo[a,h]anthracene, fluoranthene, fluorene, indeno[1,2,3-cd]pyrene, naphthalene, and phenanthrene.

The benthic risk assessment identified the following two information gaps to be filled prior to the BERA:

- Re-evaluation of TPH measures as SQVs – The uncertainties associated with any correlation of the available TPH measures (diesel- and residual-range hydrocarbons) to toxicity in the FPM have been well-documented and submitted to the EPA under separate cover (LWG 2006c). In particular, the existing TPH measures represent mixtures of hydrocarbons with an unknown, variable composition of constituents of variable toxicities and cannot be used reliably to establish an SQV. If, despite this analysis, EPA's recommendation is to utilize a TPH correlation in the BERA (in the FPM or other approach), it is critical that appropriate data be collected and that a scientifically robust analysis addressing the merits of this application be completed. The alternative to TPH would most likely be one or more PAH exposure metrics, for which we do not anticipate additional data collection. Specific metrics would be selected in collaboration with EPA (as part of the Scientific/Management Decision Point for Step 4 in the Ecological Risk Assessment Guidance for Superfund (ERAGS) Ecological Risk Assessment Process (EPA 1997a)).
- Performing a TRV search for benthic invertebrates - The TRVs used in the tissue-residue LOEs (both empirical and predicted) were based on the aquatic TRVs provided by EPA. The majority of these TRVs are derived as AWQCs multiplied with BCFs and therefore best suited as conservative screening values. A more complete TRV search for benthic invertebrates than the PRE search should be performed as part of the BERA.

8.2 FISH

The following section presents the iCOCs and potential data needs identified for fish in the LWR as determined by this Round ERA.

8.2.1 Summary of Round 2 Risk Conclusions

Multiple LOEs were used to assess risks to selected fish receptors (Section 4), including the following: 1) critical tissue-residue assessment, 2) dietary assessment, 3) surface water assessment, including a screening analysis of the potential effects of dissolved metals (i.e., copper) on salmonid olfactory function, 4) TZW assessment, and 5) a screening evaluation of sediment chemistry data relative to sediment quality guidelines derived from consideration of the lesion occurrence of skin and liver lesions associated with direct exposure of benthic fish receptors to PAHs in sediment.

Potential localized risks to fish from four iCOCs based on the empirical tissue-residue analysis and based on small home range (i.e., less than the Study Area) fish receptors: PCBs, BEHP, DDTs, TBT, and mercury. Smallmouth bass tissue composites indicated potential localized areas of risks to PCBs and BEHP, and sculpin tissue composites indicated potential localized areas of risks to DDTs and BEHP. Northern pikeminnow tissue composites indicated potential risks to mercury; however, mercury is a known to be elevated throughout the Willamette River. TBT was identified as an iCOC for fish with high uncertainty due to limited toxicological data. Cyanide, perchlorate, and 16 individual PAHs[56] were retained as potential iCOCs for fish. Potential iCOCs for fish were identified as those chemicals that were identified by the TZW LOE, but not supported by other LOEs, as posing risks to fish. These potential iCOCs are not expected to pose significant risks to fish because there is limited potential for exposure.

The primary uncertainties associated with the risk characterization for all fish receptors are whether the assumptions used (including the relevant exposure scales evaluated and assigned dietary prey portions) and the selected TRVs are appropriate for the BERA. NOAEL TRVs that were extrapolated using uncertainty factors may overestimate risks to fish. Fish iCOCs (i.e., TBT) identified using TRVs selected from a limited toxicity dataset are highly uncertain and may overestimate or underestimate risks to fish. Conservative assumptions used to evaluate fish receptors (e.g., use of single tissue composite concentrations to determine risks to selected fish species) may overestimate risks to fish; while other assumptions (i.e., assigned dietary prey portions) may overestimate or underestimate risks to fish. There is uncertainty associated with the TZW assessment based on the selection of TZW sampling sites at potential source areas and whether TZW data represent appropriate ecological exposure for fish, which may result in an overestimate of risk. No LWG-collected tissue data were available for assessing risks to lamprey ammocoetes and pre-breeding sturgeon in Round 1 or Round 2 sampling, and the risks to these selected fish receptors are unknown.

Five fish iCOCs (i.e., PCBs, DDTs, BEHP, mercury, and TBT) will be assessed and evaluated further in the BERA. The ecological relevance of the potential iCOCs (based on TZW data) to fish should be further evaluated. Estimates of site-wide risks based on a dietary exposure scenario suggest that several metals (i.e., cadmium and copper) may also contribute to fish risks, and these COPCs should be further evaluated in the BERA. All other Round 2 COPCs were not identified as iCOCs or recommended to be carried forward to the BERA because risks to fish are not expected. iPRGs were developed for all fish iCOCs, for those fish iCOCs where a tissue concentration to sediment concentration relationship could be established. iPRGs were used to identify iAOPCs for fish. Maps of iAOPCs and details of the

[56] The following PAHs were identified as potential iCOCs for fish: 2-methylnaphthalene, acenaphthene, anthracene, benzo[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene, benzo[g,h,i]perylene, benzo[k]fluoranthene, chrysene, dibenzo[a,h]anthracene, fluoranthene, fluorene, indeno[1,2,3-cd]pyrene, naphthalene, phenanthrene, and pyrene.

methods for developing iAOPCs are presented in Section 10 of the Comprehensive Round 2 Report.

8.2.2 Potential Data Needs

The following sampling needs, identified by EPA and its partners, for the fish assessment are being conducted as part of the Round 3 sampling:

- Collection of lamprey ammocoete tissue for chemical analysis (conducted in fall 2006) and comparison to tissue-based TRVs.
- Conduct lamprey ammocoete toxicity tests (sampling event initiated in fall 2006) to determine appropriate toxicological thresholds.
- Collection of pre-breeding sturgeon tissue for chemical analysis (to be conducted in winter/spring 2007) and comparison to tissue-based TRVs. This analysis will include the analysis of TBT in whole-body tissue. Observations on external/physical conditions of pre-breeding sturgeon collected in the field will be made.

Additional surface water collected as part of Round 3 (currently being conducted) will be incorporated into the fish assessment of the BERA to better understand any influence of flow conditions on surface water concentrations.

In addition to Round 3 sampling needs, the following information gaps were identified:

- Round 2 fish TRVs will be re-evaluated to determine whether they are appropriate for the BERA.
- Additional literature investigations will be conducted as part of the BERA to evaluate the significance of exposure to individual iCOCs to fish populations.

8.3 WILDLIFE

The following section presents the iCOCs and potential data needs identified for wildlife in the LWR as determined by this Round ERA.

8.3.1 Summary of Round 2 Risk Conclusions

Multiple LOEs were used to assess risks to selected wildlife receptors (Section 5), including the dietary dose assessment and the bird egg tissue assessment. The wildlife iCOCs identified in the Round 2 ERA are PCBs, dioxins and furans, mercury, DDTs (including sum DDD and sum DDE), and aldrin. Study Area-wide risks to wildlife receptors were identified based on dietary exposure to PCBs and dioxins and furans. DDE represented a Study Area-wide risk to osprey and bald eagle based on the bird

egg tissue LOE. Potential localized risks to shorebirds (represented by spotted sandpiper) were associated with six specific foraging beach areas and were based on dietary exposure to PCBs, dioxins and furans, sum DDD, and aldrin; however, the shorebird risk analysis is based on highly conservative assumptions and may overestimate risks to shorebirds.

The primary uncertainties associated with the risk characterization for wildlife are whether the dietary assumptions used (including site use factors and the spatial scale evaluation) and the selected TRVs are appropriate for the BERA. NOAEL TRVs that were extrapolated using uncertainty factors may overestimate risks to wildlife. Bird egg TRVs based on field studies and dietary dose TRVs extrapolated using uncertainty factors may overestimate risks to wildlife. Most of the assumptions used to evaluate wildlife receptors were highly conservative (e.g., spotted sandpipers forage and breed from within only the habitat beach areas; the sandpiper diet consists only of laboratory-exposed worms) and may overestimate risks to wildlife. However, the assumption that bald eagles consume only fish may underestimate risks from specific iCOCs (i.e., PCBs). There is additional uncertainty associated with the field-based biomagnification factors used to estimate bird egg tissue concentrations from fish tissue concentrations, which may overestimate or underestimate risks.

Five wildlife iCOCs (i.e., PCBs, dioxins and furans, DDTs [as sum DDE and sum DDD], and aldrin) will be assessed and evaluated further in the BERA. All other Round 2 COPCs were not identified as iCOCs or recommended to be carried forward to the BERA because risks to wildlife are not expected. iPRGs were developed for all wildlife iCOCs in order to identify iAOPCs. Maps of iAOPCs and details of the methods for developing iAOPCs are presented in Section 10 of the Comprehensive Round 2 Report.

8.3.2 Potential Data Needs

No additional sampling data needs were identified for wildlife. However, the following assumptions used in the model will be further evaluated for the BERA to reduce uncertainties in risk estimates:

- Evaluation of appropriate site use factors and spatial scale assumptions for selected wildlife receptors. Additional literature will be reviewed to determined appropriate site use factor and spatial scale assumptions to be used in the BERA.
- Further evaluation of the ecological relevant of shorebird exposure assumptions (i.e., shorebird beach habitat use in the Study Area, the use of laboratory-exposed worms to represent shorebird prey).
- Round 2 wildlife TRVs will be re-evaluated to determine whether they are appropriate for the BERA.

- The evaluation of direct measurement regional data on wildlife populations of our selected ecological receptors in the LWR, if such data are available.

8.4 AMPHIBIANS/REPTILES AND AQUATIC PLANTS

The following section presents the conclusions of the amphibian/reptile and aquatic plant assessment as determined by this Round ERA.

8.4.1 Summary of Round 2 Risk Conclusions

One LOE, a surface water assessment, was used to assess risks to amphibians/reptiles (Sections 6.0), and two LOEs, surface water and TZW assessments, were used to assess risks to aquatic plants (Section 7). No iCOCs were identified for either amphibians/reptiles or aquatic plants. Because no iCOCs were identified for these receptor groups, no iAOPCs were developed. These receptor groups will be summarized in the BERA; however no further analysis will be conducted.

8.4.2 Potential Data Needs

Additional surface water collected as part of Round 3 (currently being conducted) will be incorporated into an updated screening assessment and analysis for amphibians/reptiles and aquatic plants in the BERA. This analysis will be conducted to better understand any influence of flow conditions on surface water concentrations.

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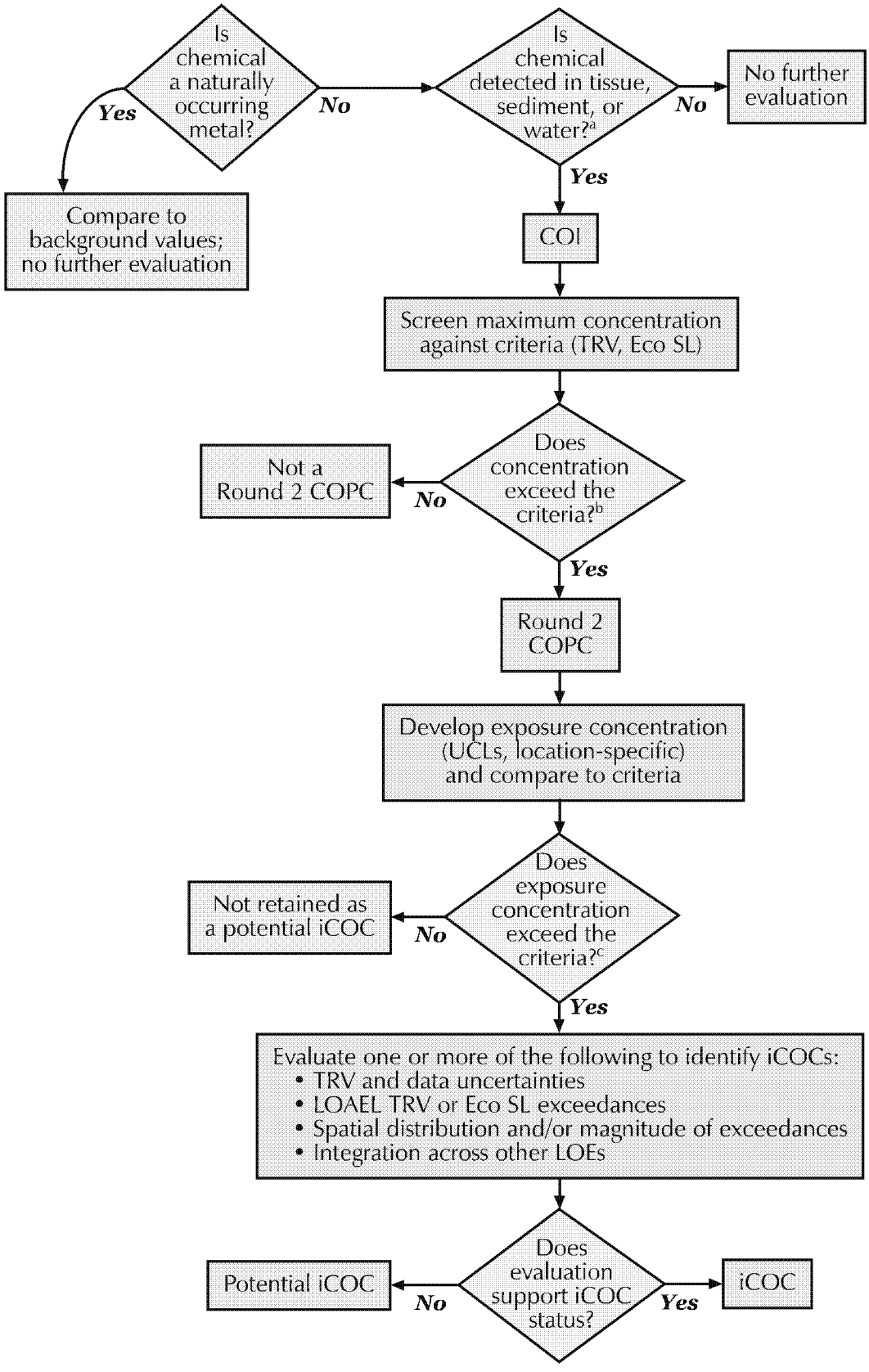

[a] For the benthic predicted toxicity LOE, COIs were identified as chemicals detected in more than 30 and 100 sediment samples for the FPM and LRM, respectively.

[b] For the benthic predicted toxicity LOE, COPCs were identified as chemicals associated with toxicity in either of the two models.

[c] For the benthic predicted toxicity LOE, potential iCOCs were identified as those chemicals associated with toxicity in the FPM.

Figure 2-1
Process for Identification of Ecological iCOCs

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Appendix G

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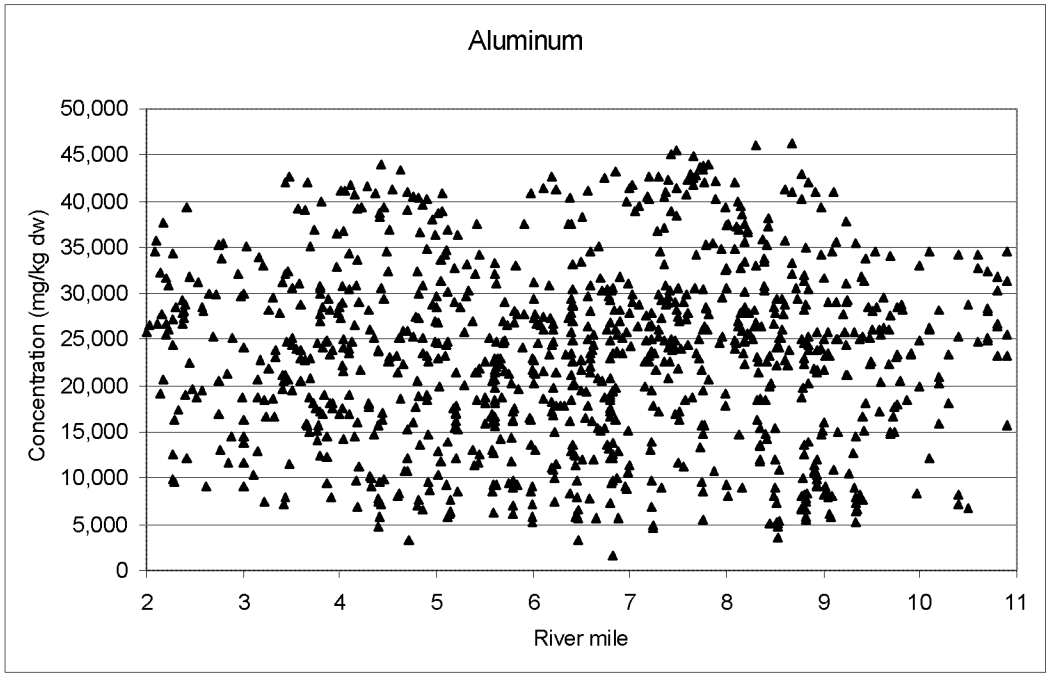


Figure 2-2
Aluminum Concentrations in Sediment by River Mile

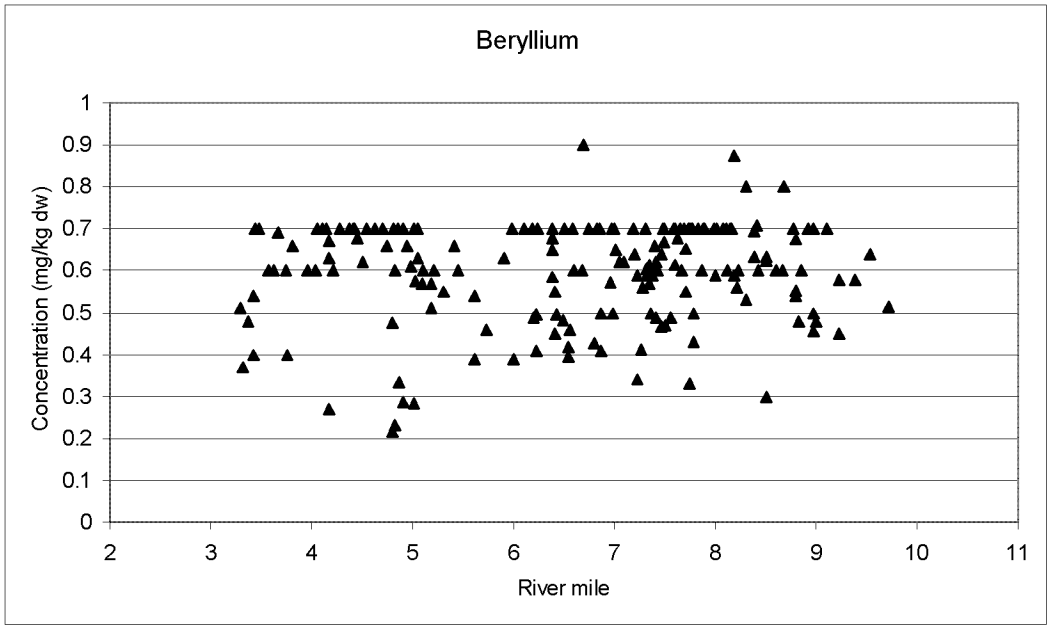


Figure 2-3
Beryllium Concentrations in Sediment by River Mile

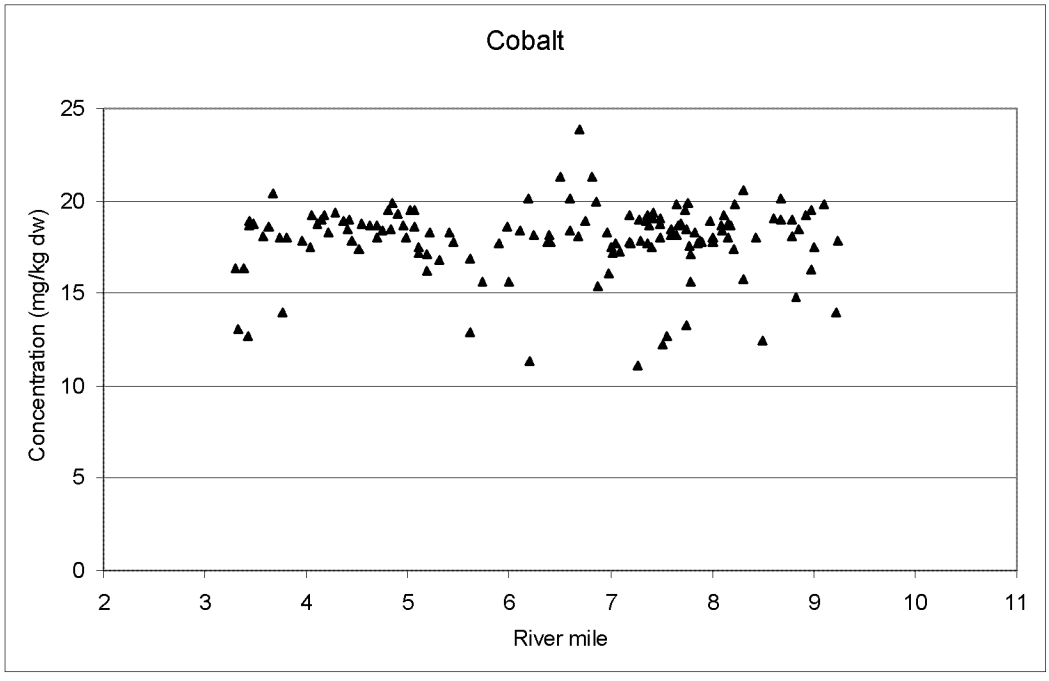


Figure 2-4
Cobalt Concentrations in Sediment by River Mile

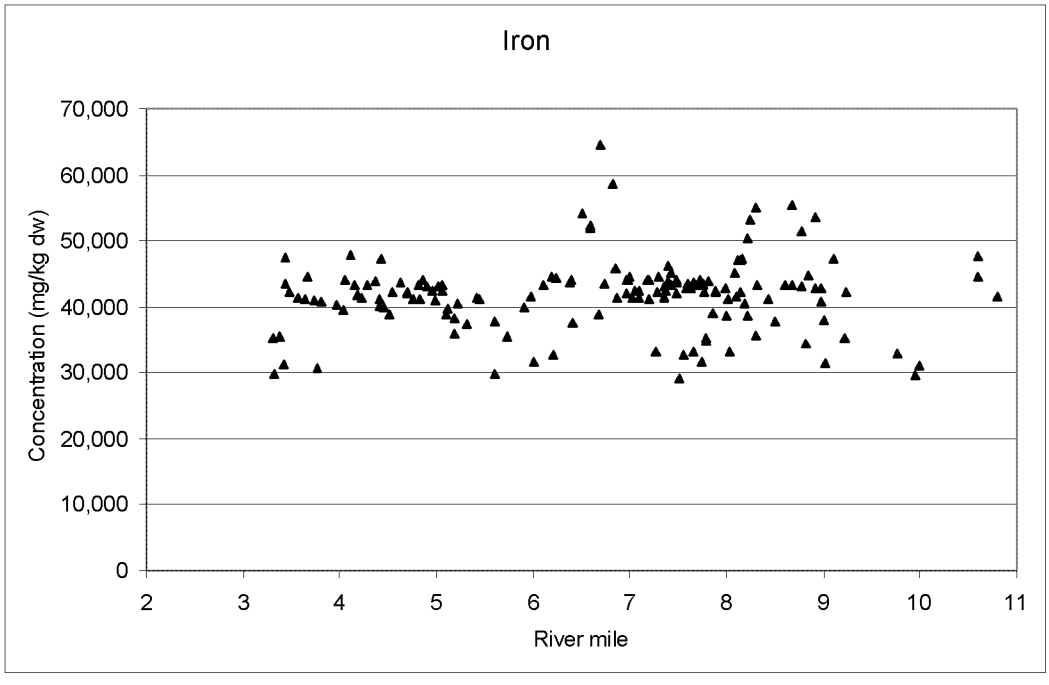


Figure 2-5
Iron Concentrations in Sediment by River Mile

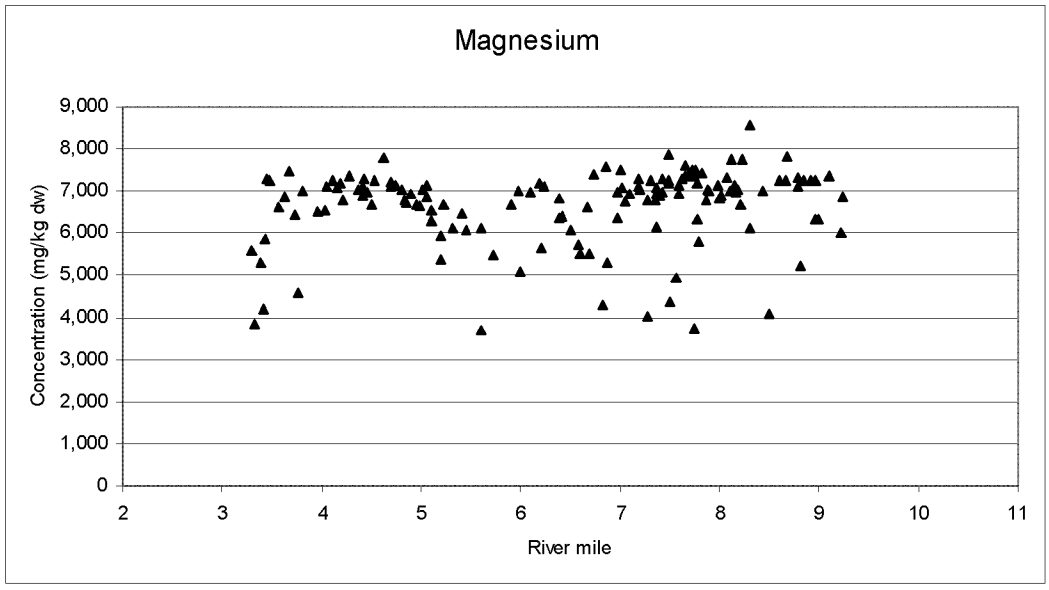


Figure 2-6
Magnesium Concentrations in Sediment by River Mile

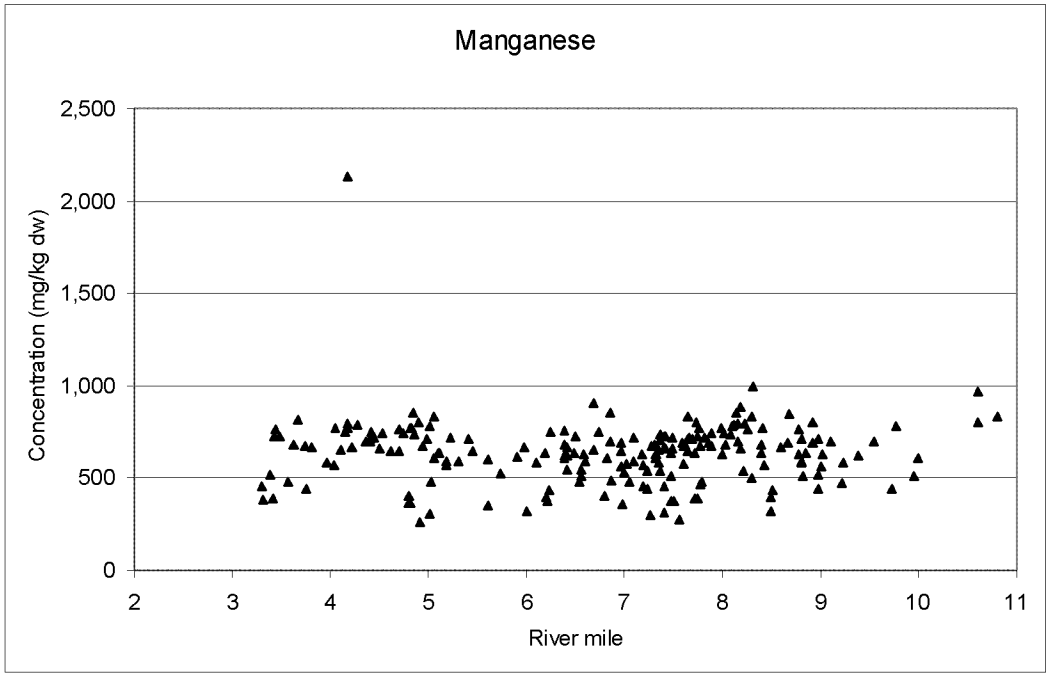


Figure 2-7
Manganese Concentrations in Sediment by River Mile

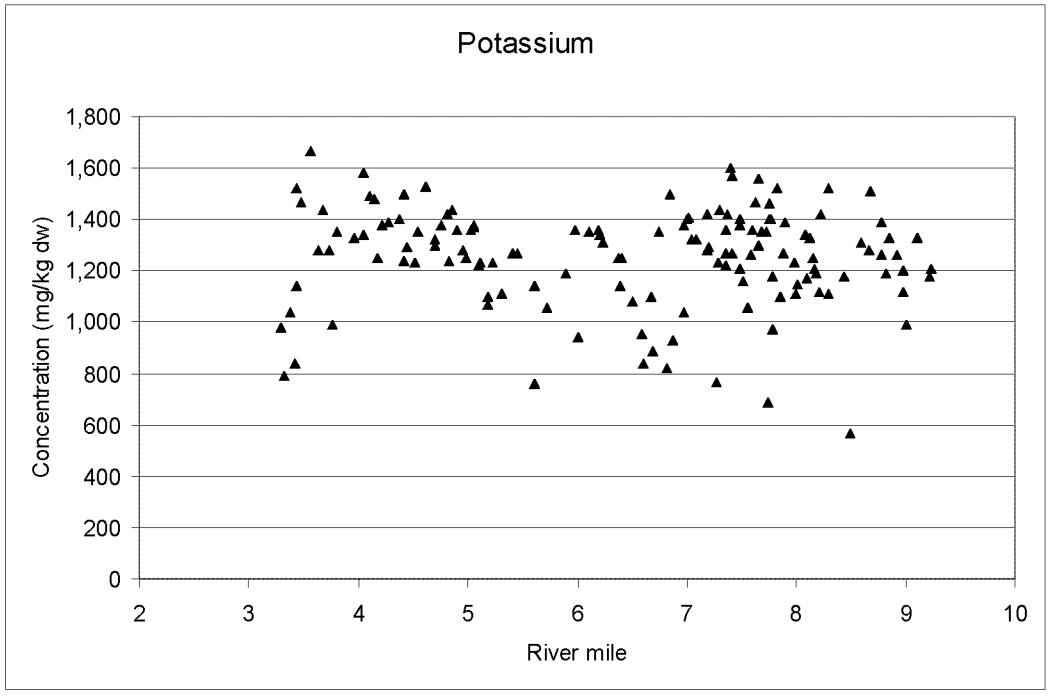
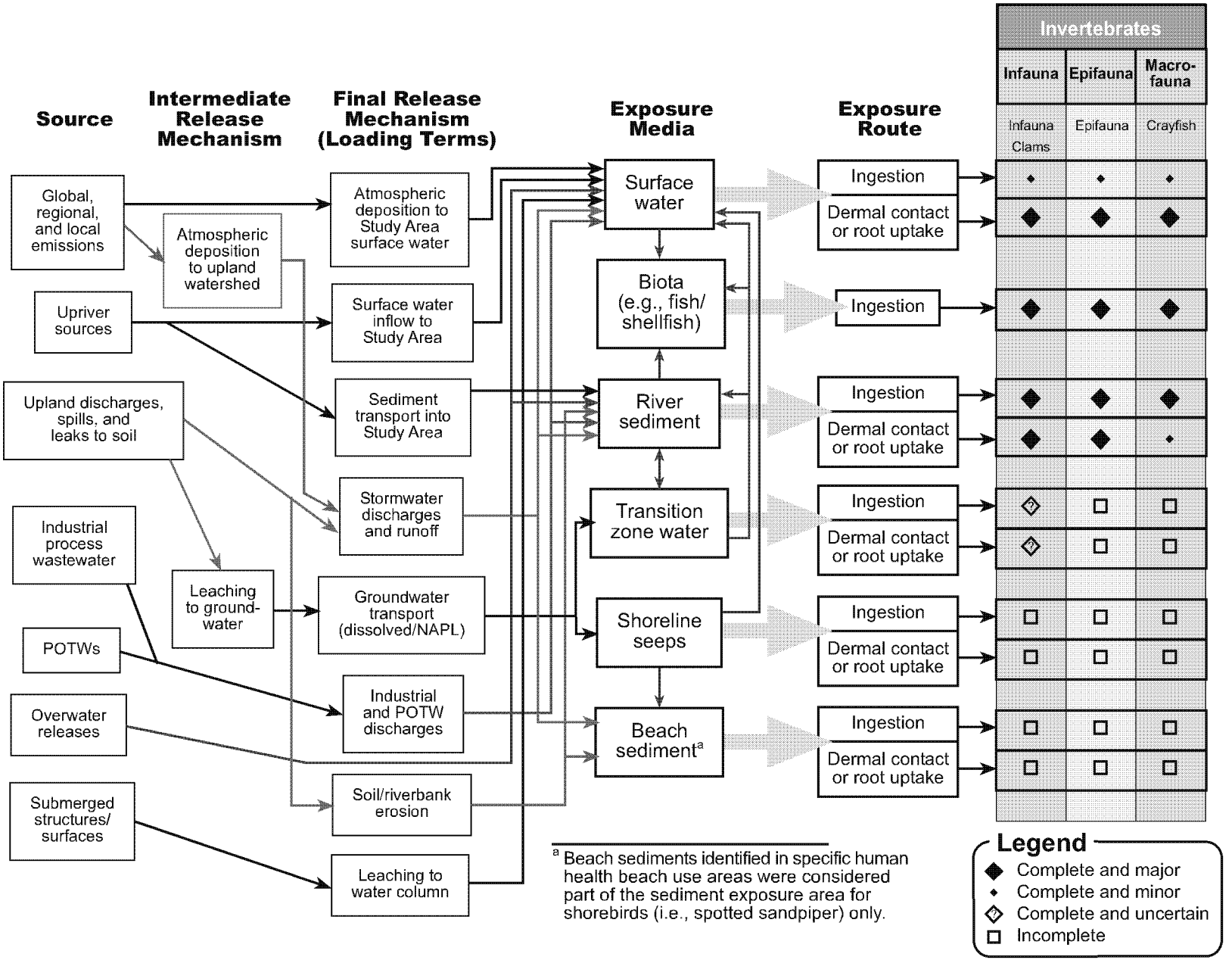


Figure 2-8
Potassium Concentrations in Sediment by River Mile



Wind Ward LLC environmental

LWG Lower Willamette Group

Figure 3-1
Preliminary Ecological Conceptual Site Model - Benthic Invertebrates

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Figure 3-2
Invertebrate Tissue Sampling and Sediment Sampling Locations for Bioaccumulation Tests

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Appendix G

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Prepared by STS, 10/30/06, Modified by AMW 01/29/07, Map 2632, W:\Projects\01-25-01 Portland Harbor\ERA\Data\GIS\Events\BenthicRisk

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N

0 1,000 2,000 3,000 Feet

Upland site
Waterfront tax lot
Arterial
Highway
Freeway
Railroad
Dock or structure
River mile
Navigation channel
13-foot contour (in-river analysis elev.)
River or slough

FEATURE SOURCES:
Transportation, Property, or Boundaries: Metro RLIS .
Channel & River miles: US Army Corps of Engineers.

Figure 3-3
Near-Bottom Surface Water Sampling Locations

Portland Harbor RI/FS
Comprehensive Round 2 Report
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BZTO104(e)030697

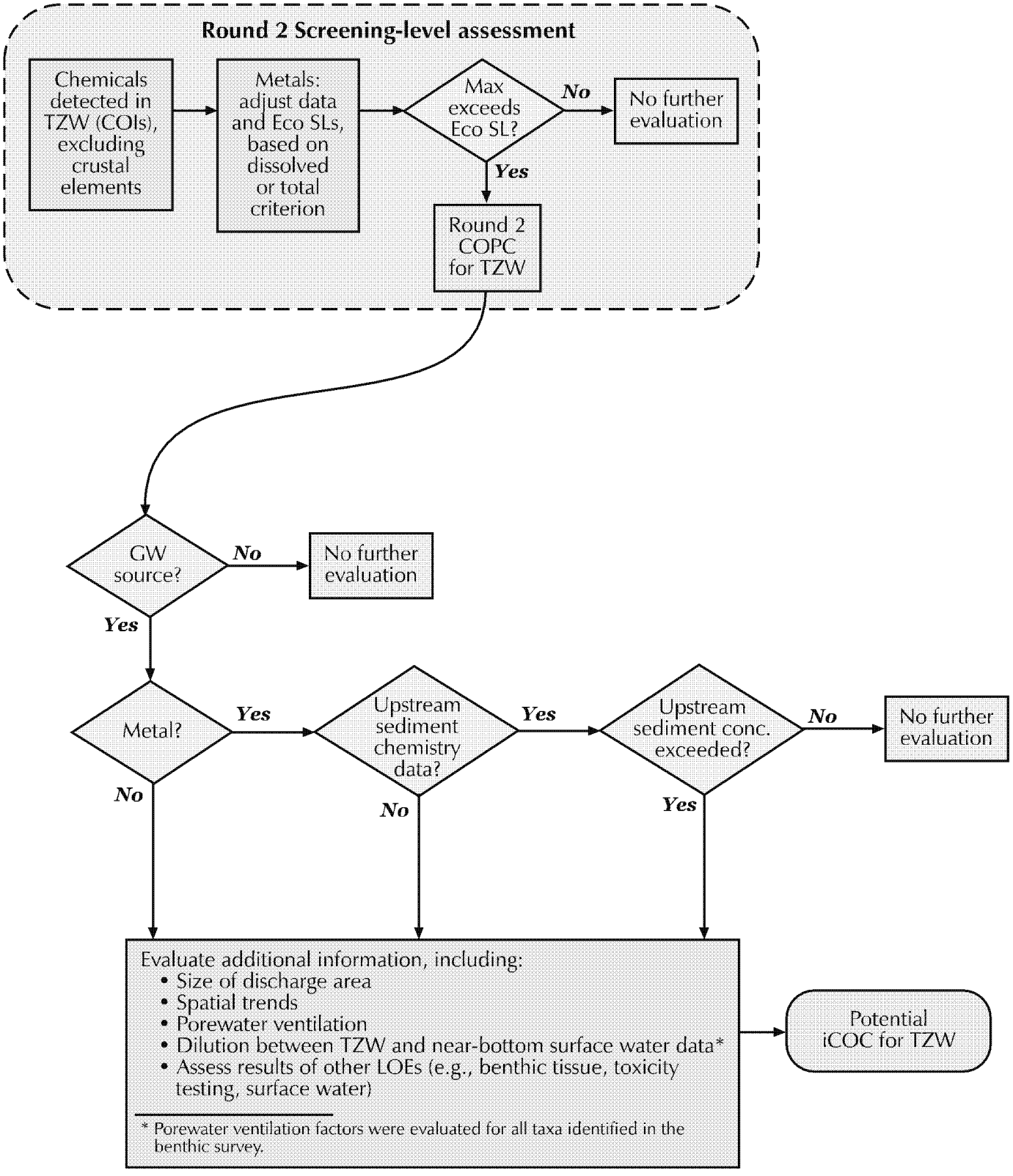

Figure 3-4
TZW Evaluation Framework for Ecological Receptors

Portland Harbor RI/FS
Comprehensive Round 2 Report
Appendix G

LWG Lower Willamette Group

Windward environmental LLC

Prepared by STS 10/3/06. Modified by AMW 01/29/07. MAP 2636. W:\Projects\01-25-01 Portland Harbor ERA\Data\GIS\Events\TZW

Wind Ward LLC environmental

LWG Lower Willamette Group

Figure 3-5
Transition Zone Water Sampling Locations
River Miles 2 to 11

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Comprehensive Round 2 Report
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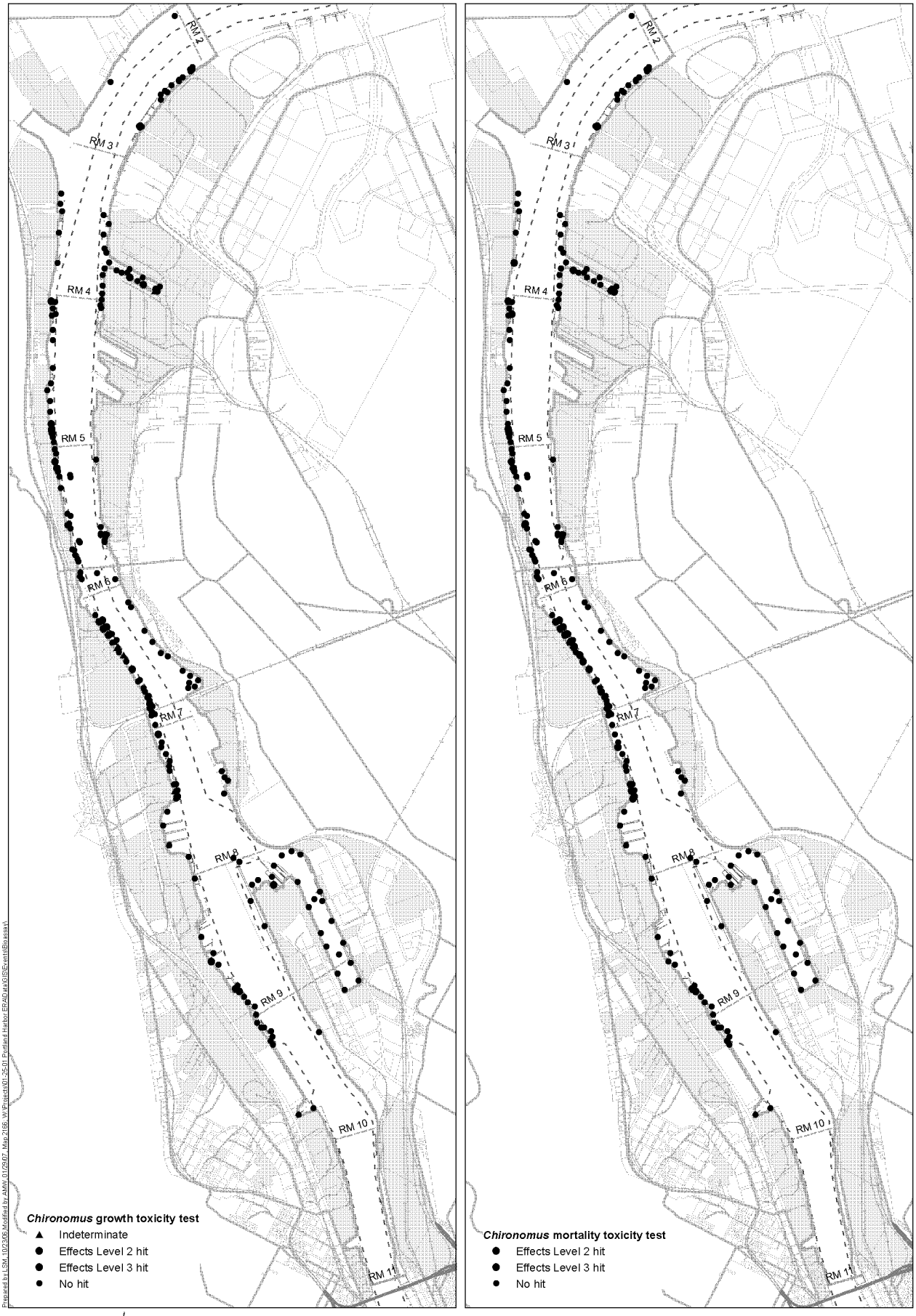
BZTO104(e)030699



Prepared by LSM, 10/23/06 Modified by AMW 01/29/07, Map 2166, W:\Projects\01-25-01 Portland Harbor EPA\Data\GIS\Events\Bioassay\

Wind Ward LLC environmental

LWG LOWER WILLAMETTE GROUP

N

0 1,000 2,000 3,000 4,000 Feet

Upland site
Waterfront tax lot
Arterial
Highway
Freeway
Railroad
Dock or structure
River mile
Navigation channel
13-foot contour (in-river analysis elevation)
River or slough
Capping since Round 2 field work
Dredging since Round 2 field work

FEATURE SOURCES:
Transportation, Property, or Boundaries: Metro RLIS.
Channel & River miles: US Army Corps of Engineers.

Figure 3-6
Toxicity Test Hit, No-Hit, and Indeterminate Stations for *Chironomus* River Miles 2 to 11

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Comprehensive Round 2 Report
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Prepared by LSM, 10/23/06. Modified by AMW 01/29/07. Map 2166. W:\Projects\01-2501 Portland Harbor ERA\Data\GIS\Events\fig3-7.aprx\

Wind Ward environmental LLC

LWG Lower Willamette Group

N

0 1,000 2,000 3,000 4,000 Feet

Upland site
Waterfront tax lot
Arterial
Highway
Freeway
Railroad
Dock or structure
River mile
Navigation channel
13-foot contour (in-river analysis elevation)
River or slough
Capping since Round 2 field work
Dredging since Round 2 field work

FEATURE SOURCES:
Transportation, Property, or Boundaries: Metro RLIS.
Channel & River miles: US Army Corps of Engineers.

Figure 3-7
Toxicity Test Hit, No-Hit, and Indeterminate Stations for *Hyalella*
River Miles 2 to 11

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Comprehensive Round 2 Report
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Prepared by LSM 10/30/06. Modified by AMW 01/29/07. MAP 2607. W:\Projects\01-2501 Portland Harbor ERA\Data\GIS\Events\Bioassay\AOPCs

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Figure 3-8
Toxicity Test Results and FPM Prediction of Benthic Toxicity
River Miles 2 to 11

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Prepared by LSM 10/24/06; Modified by AMW 01/29/07. MAP 2608. W:\Projects\01-25-01 Portland Harbor ERA\Data\GIS\Events\Bioassay\AOPCs

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Figure 3-9
Toxicity Test Results and LRM Predictions of Benthic Toxicity
River Miles 2 to 11

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Comprehensive Round 2 Report
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Prepared by LSM 11/02/06. Modified by AWW 01/29/07. MAP 2644. W:\Projects\01-2501 Portland Harbor ERA\Data\GIS\Events\Biassa\ModelComparison

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Figure 3-10
Toxicity Test Results and Combined LRM and FPM Predictions of Benthic Toxicity
River Miles 2 to 11

Portland Harbor RI/FS
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Wind Ward environmental LLC

LWG Lower Willamette Group

N

0 1,000 2,000 3,000 Feet

Upland site
Waterfront tax lot
Arterial
Highway
Freeway
Railroad
Dock or structure
River mile
Navigation channel
13-foot contour (in-river analysis elev.)
River or slough

FEATURE SOURCES:
Transportation, Property, or Boundaries: Metro RLIS.
Channel & River miles: US Army Corps of Engineers.

Figure 3-11
Field-Collected Tissue with HQs > 1.0

Portland Harbor RI/FS
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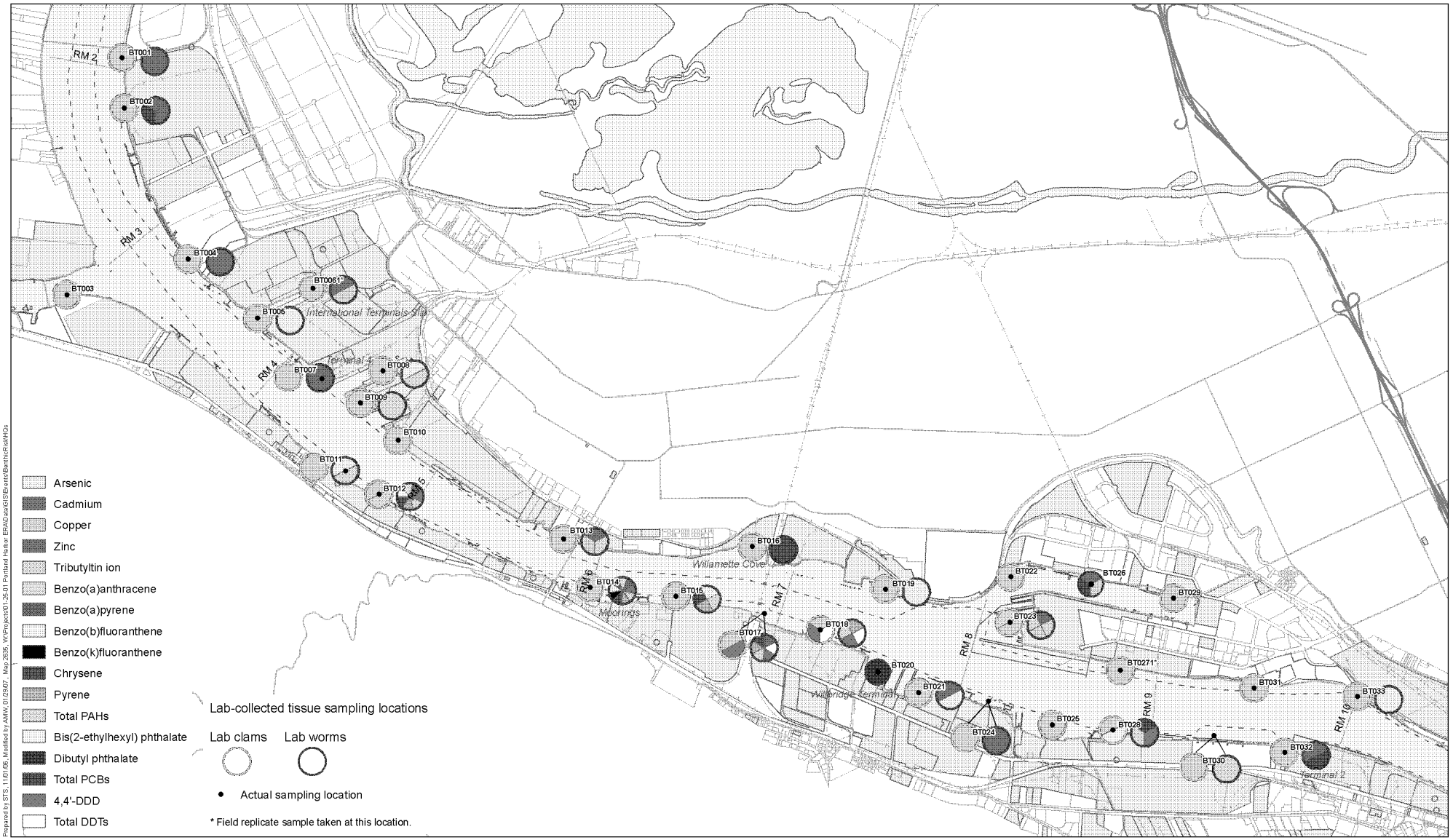
BZTO104(e)030705



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LWG Lower Willamette Group

N

0 1,000 2,000 3,000 4,000 5,000 Feet

Upland site
Waterfront tax lot
Arterial
Highway
Freeway
Railroad
Dock or structure
River mile
Navigation channel
13-foot contour (in-river analysis elev.)
River or slough

FEATURE SOURCES:
Transportation, Property, or Boundaries: Metro RLIS.
Channel & River miles: US Army Corps of Engineers.

Figure 3-12
Laboratory-Collected Tissue with HQs > 1.0

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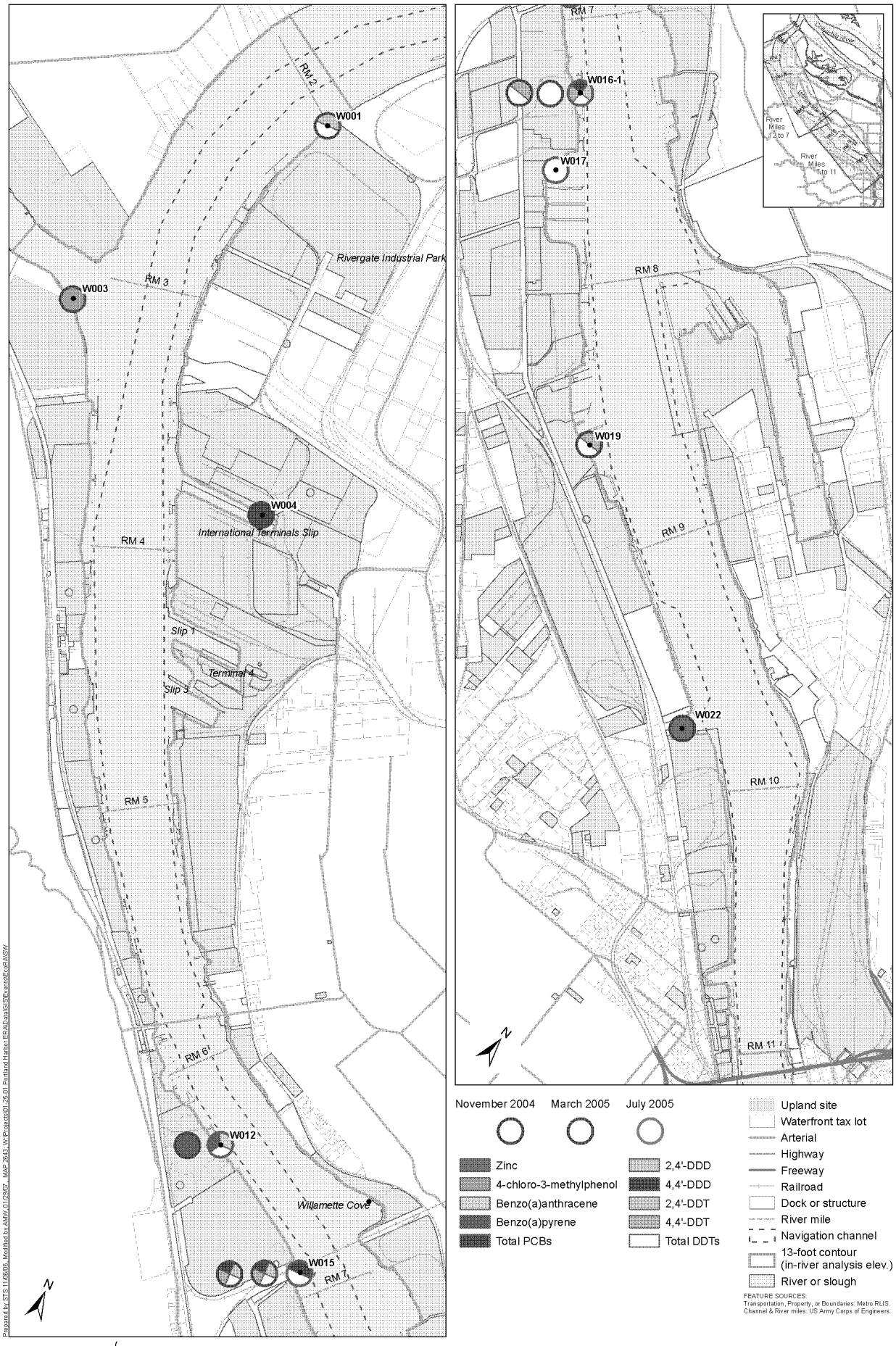
BZTO104(e)030706



Prepared by STS 11/06/06, Modified by AMW 01/23/07 MAP 2643 W:\Projects\01-25-01 Portland Harbor ERA\Data\GIS\Events\Eco\RASW

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Figure 3-13
Round 2 Near-Bottom Surface Water COPCs with Chronic HQs > 1.0
River Miles 2 to 11

Portland Harbor RI/FS
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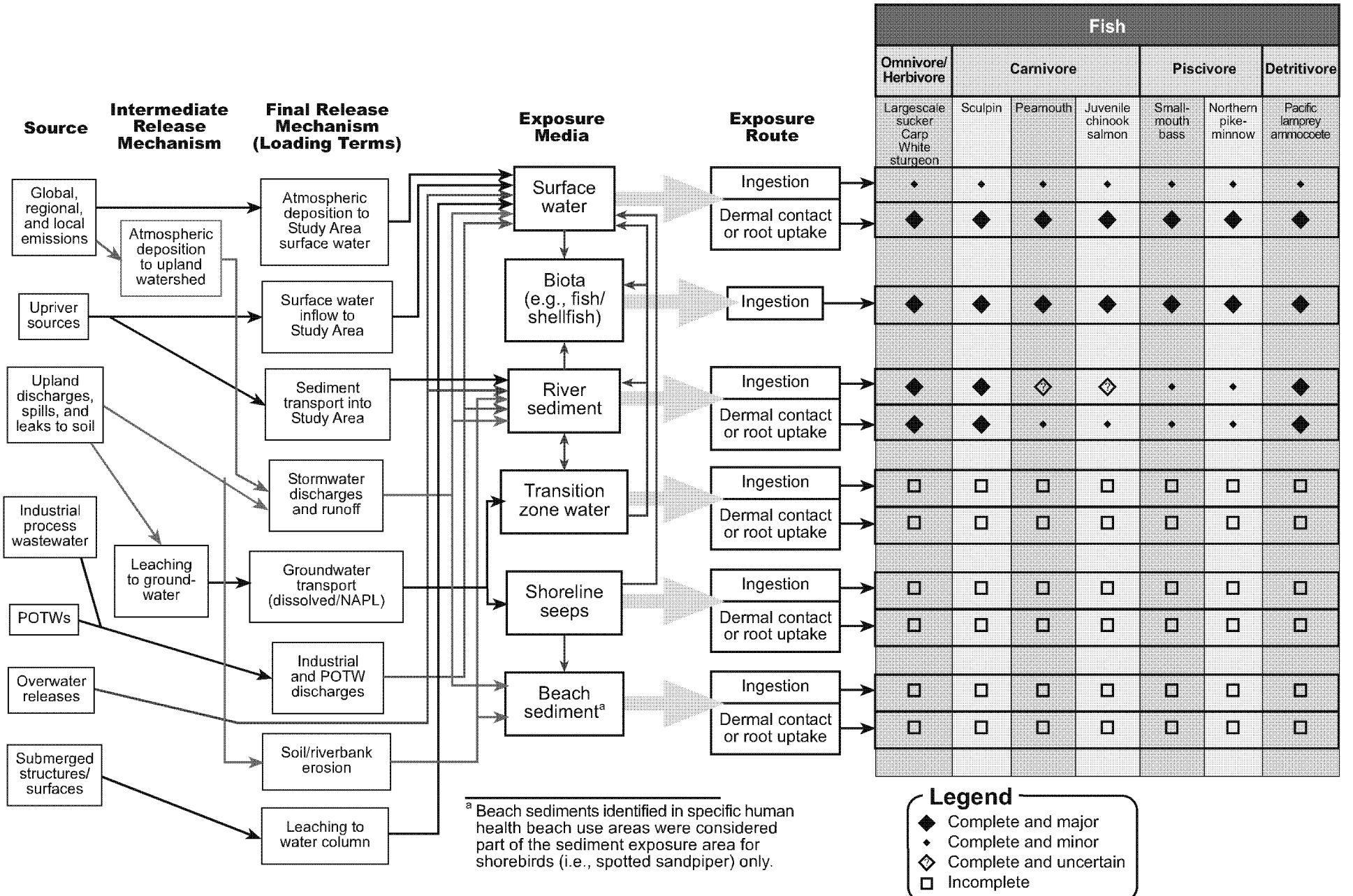

[a] Beach sediments identified in specific human health beach use areas were considered part of the sediment exposure area for shorebirds (i.e., spotted sandpiper) only.

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Figure 4-1
Preliminary Ecological Conceptual Site Model - Fish

Portland Harbor RI/FS
Comprehensive Round 2 Report
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BZTO104(e)030708

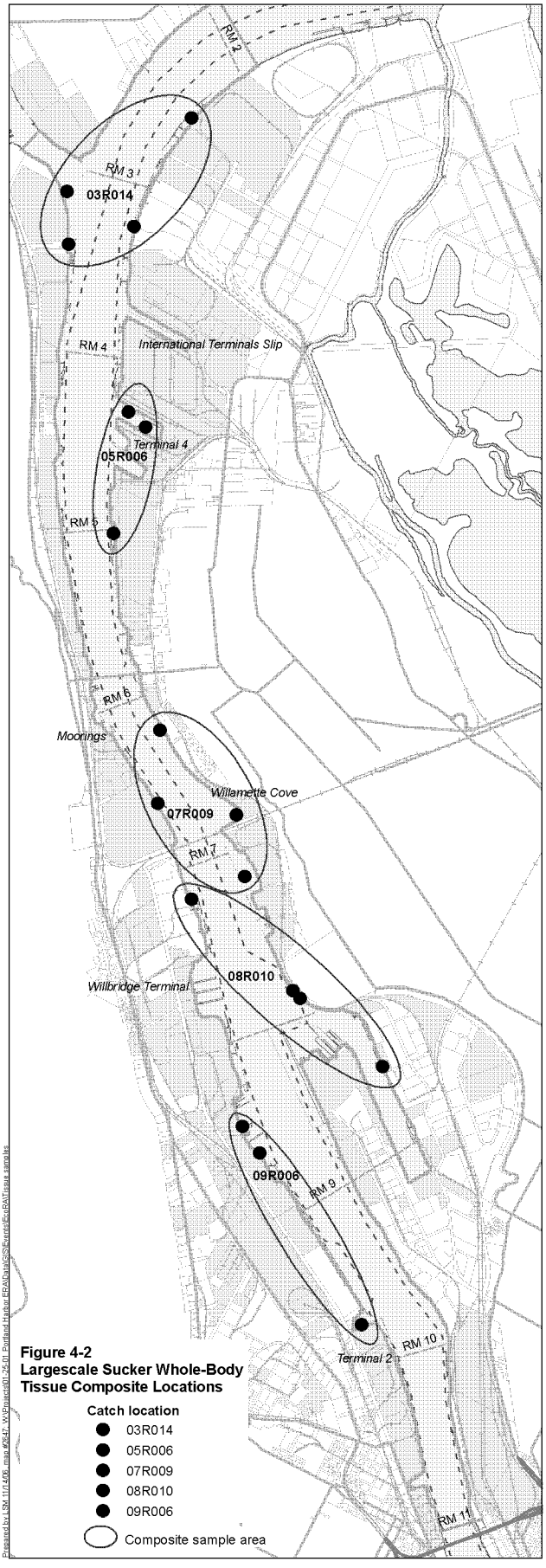

Figure 4-2
Largescale Sucker Whole-Body Tissue Composite Locations

Catch location
- 03R014
- 05R006
- 07R009
- 08R010
- 09R006
- Composite sample area

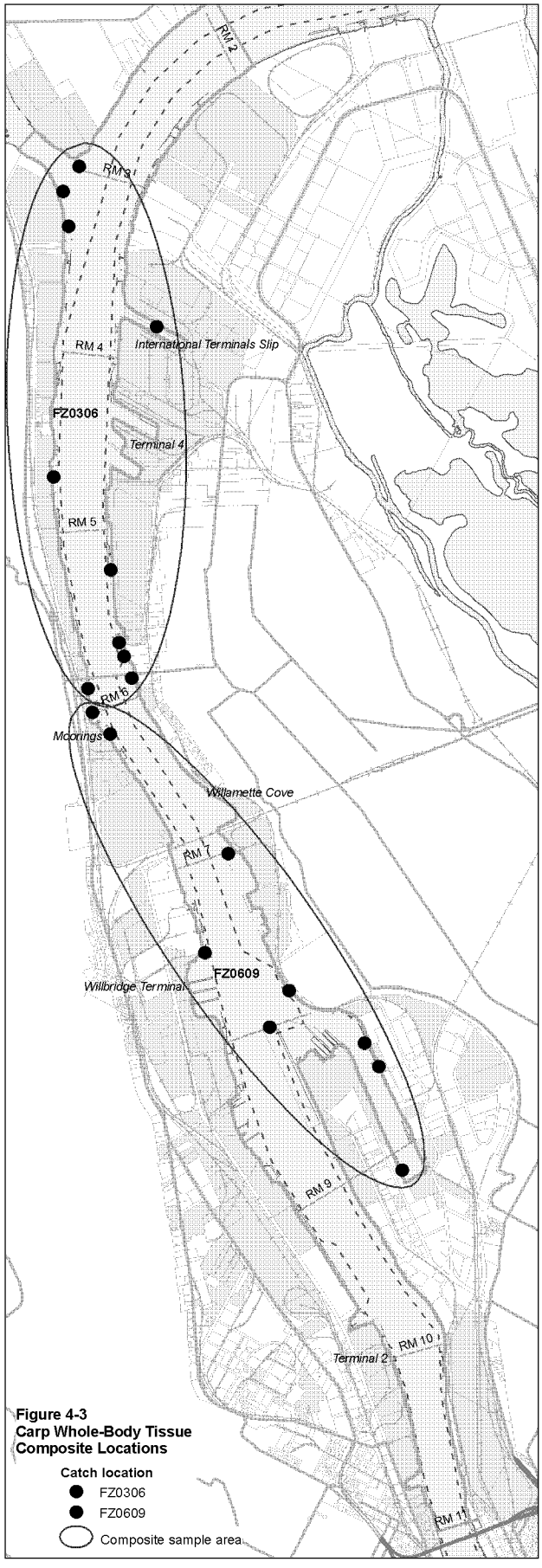

Figure 4-3
Carp Whole-Body Tissue Composite Locations

Catch location
- FZ0306
- FZ0609
- Composite sample area

Prepared by LSM 11/7/406 map #2647 W:\Projects\01-26-01 Portland Harbor ERA\Data\GIS\Events\EcoRAT\tissue samples

Wind Ward environmental LLC

LWG LOWER WILLAMETTE GROUP

N

0 1,000 2,000 3,000 4,000 Feet

Upland site
Waterfront tax lot
Arterial
Highway
Freeway
Railroad
Dock or structure
River mile
Navigation channel
13-foot contour (in-river analysis elevation)
River or slough

FEATURE SOURCES:
Transportation, Property, or Boundaries: Metro RLIS.
Channel & River miles: US Army Corps of Engineers.

Figure 4-2
Figure 4-3
River Miles 2 to 11

Portland Harbor RI/FS
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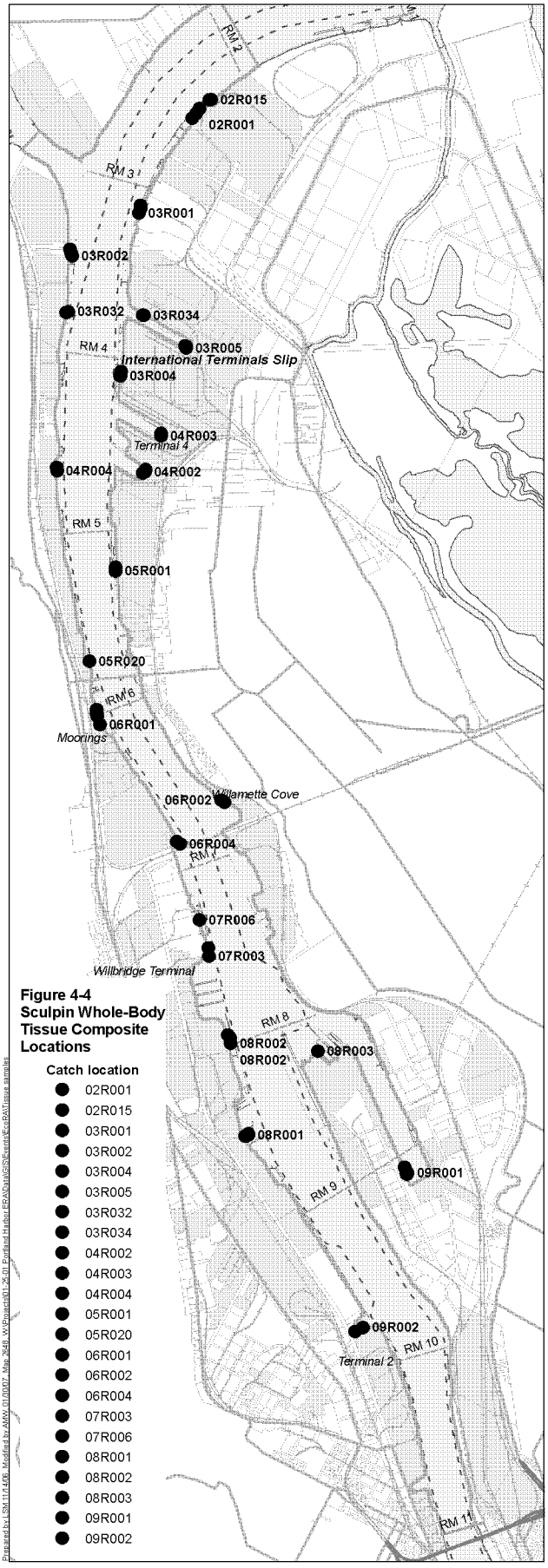

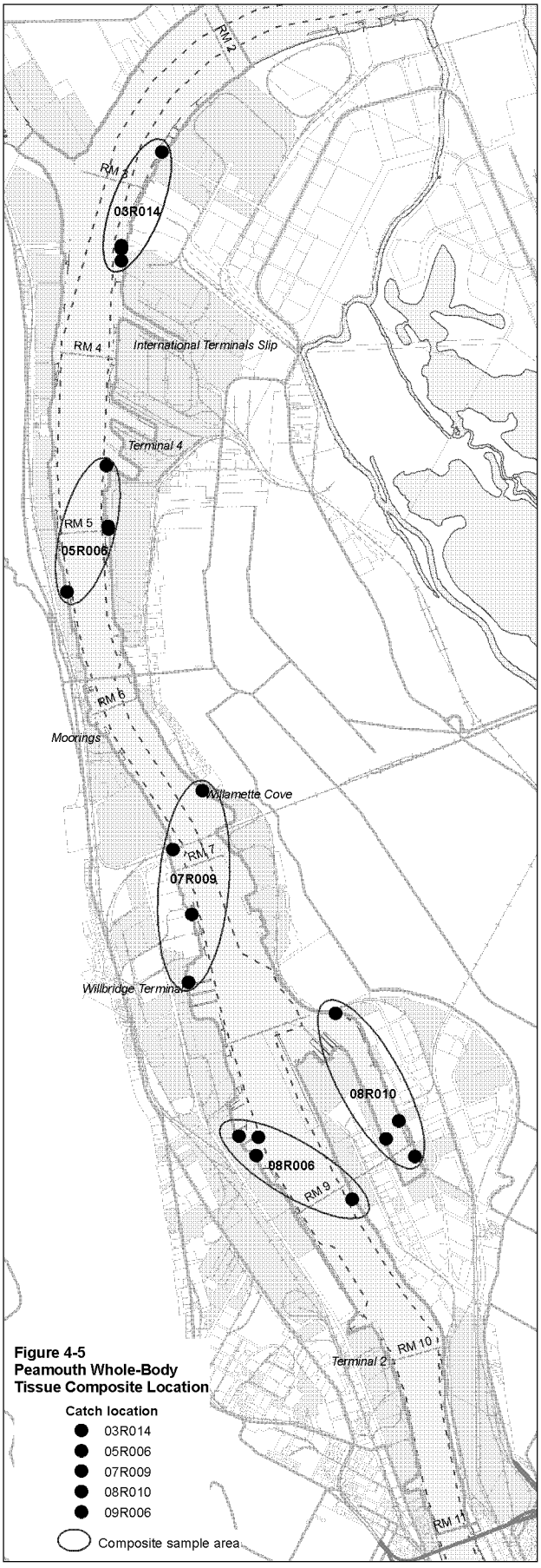

Prepared by LSM 11/7/406. Modified by AMW 01/30/07. Map: 2648. W:\Projects\01-25-01 Portland Harbor ERA\Data\GIS\Events\EcoERA\Tissue samples

Wind Ward LLC environmental

LWG LOWER WILLAMETTE GROUP

N

0 1,000 2,000 3,000 4,000 Feet

Upland site
Waterfront tax lot
Arterial
Highway
Freeway
Railroad
Dock or structure
River mile
Navigation channel
13-foot contour (on-river analysis elevation)
River or slough

FEATURE SOURCES:
Transportation, Property, or Boundaries: Metro RLIS.
Channel & River miles: US Army Corps of Engineers.

Figure 4-4
Figure 4-5
River Miles 2 to 11

Portland Harbor RI/FS
Comprehensive Round 2 Report
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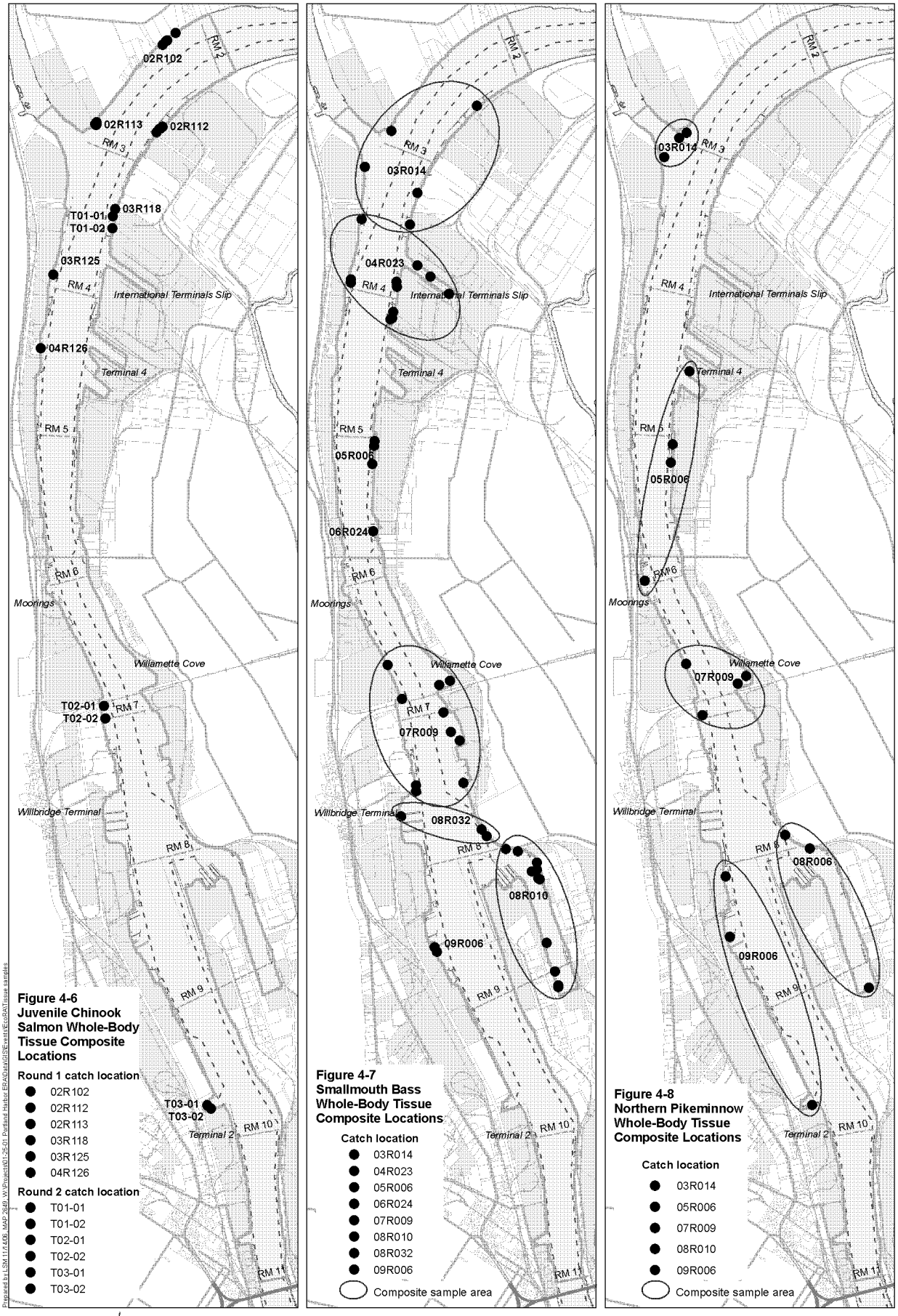

Prepared by LSM 11/7/406, MAP 2648 W:\Projects\01-25-01 Portland Harbor ERA\Data\GIS\Events\EcoRA\Tissue samples

Wind Ward LLC environmental

LWG LOWER WILLAMETTE GROUP

N

0 1,000 2,000 3,000 4,000 Feet

Upland site
Waterfront tax lot
Arterial
Highway
Freeway
Railroad
Dock or structure
River mile
Navigation channel
13-foot contour (in-river analysis elevation)
River or slough

FEATURE SOURCES:
Transportation, Property, or Boundaries: Metro RLIS.
Channel & River miles: US Army Corps of Engineers.

Figure 4-6
Figure 4-7
Figure 4-8
River Miles 2 to 11

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BZTO104(e)030711

Prepared by STS, 11/03/06, MAP 2546, W:\Projects\01-25-01 Portland Harbor ERA\Data\GIS\Events\EcoRAS\W

Wind Ward LLC environmental

LWG Lower Willamette Group

Figure 4-9
Surface Water Sampling Locations Used to Evaluate Exposure of Fish to Surface Water
River Miles 2 to 11

Portland Harbor RI/FS
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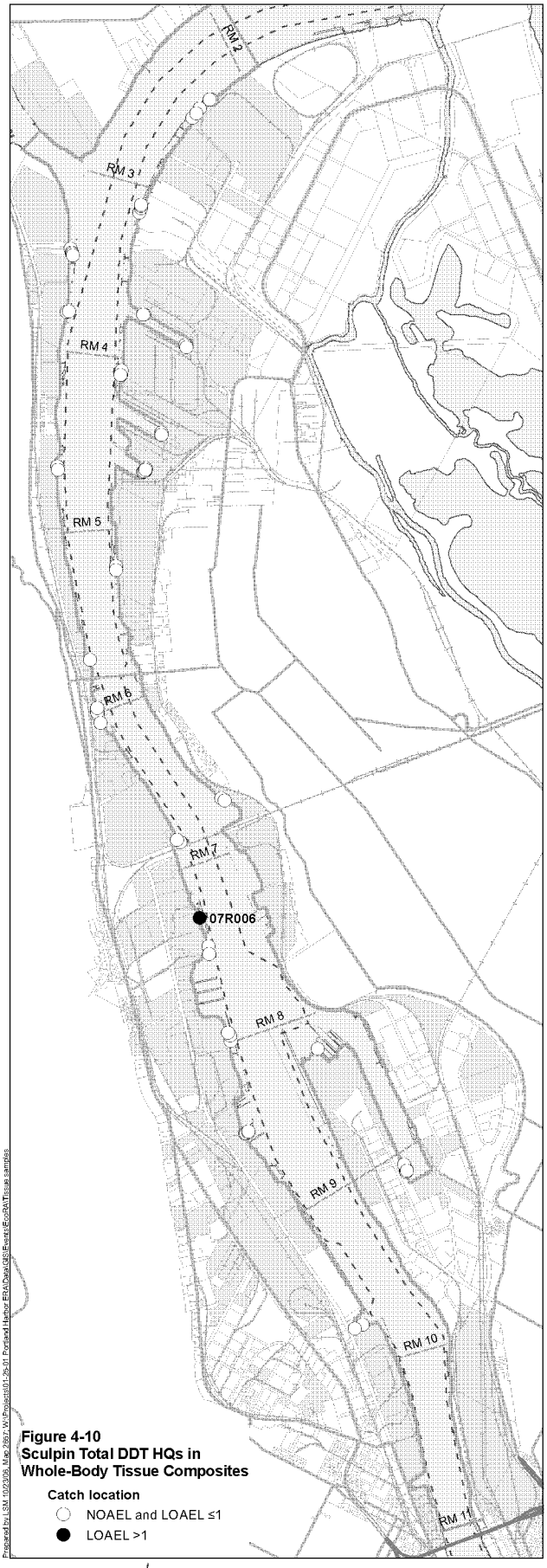

Figure 4-10
Sculpin Total DDT HQs in Whole-Body Tissue Composites

Catch location

- NOAEL and LOAEL ≤1
- LOAEL >1

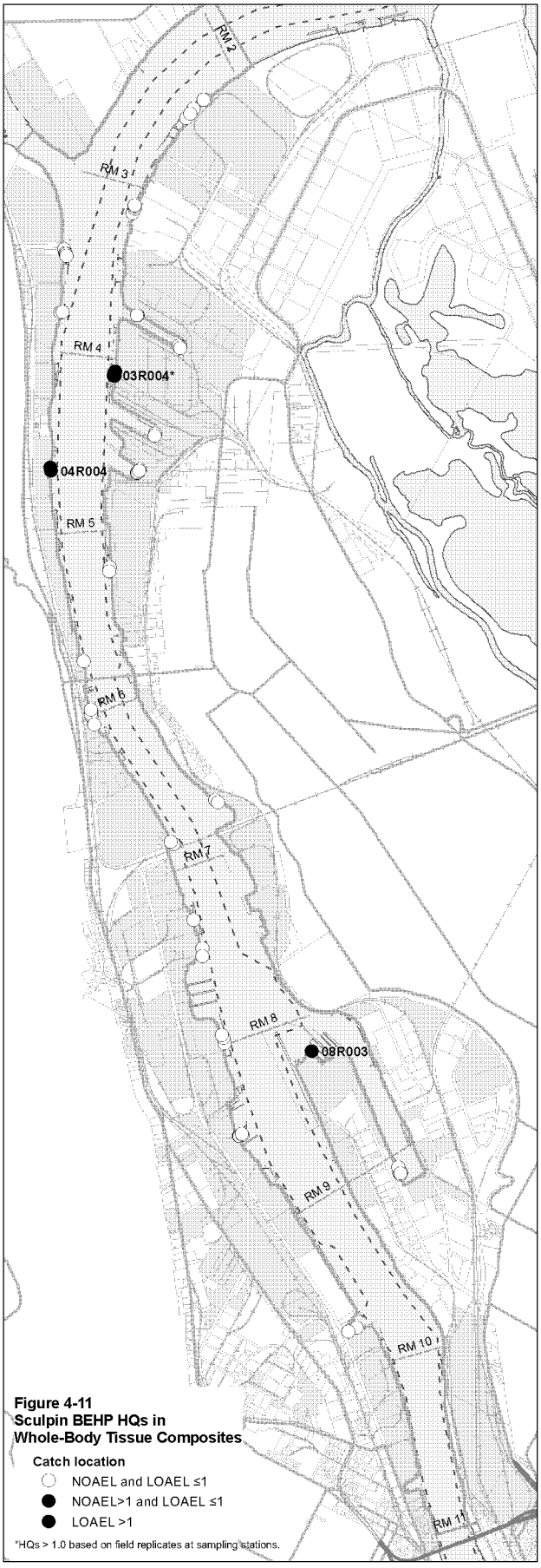

Figure 4-11
Sculpin BEHP HQs in Whole-Body Tissue Composites

Catch location

- NOAEL and LOAEL ≤1
- NOAEL>1 and LOAEL ≤1
- LOAEL >1

*HQs > 1.0 based on field replicates at sampling stations.

Prepared by LSM 10/23/06, Map 2667; W:\Projects\01-25-01 Portland Harbor ERA\Data\GIS\Bank\EcoRA\Tissue samples

Wind Ward LLC environmental

LWG LOWER WILLAMETTE GROUP

N

0 1,000 2,000 3,000 4,000 Feet

Waterfront tax lot
Upland site
Arterial
Highway
Freeway
Railroad
Dock or structure
River Mile
Navigation Channel
13-foot contour (in-river analysis elevation)
River or slough

FEATURE SOURCES:
Transportation, Property, or Boundaries: Metro RLIS .
Channel & River miles: US Army Corps of Engineers.

Figure 4-10
Figure 4-11
River Miles 2 to 11

Portland Harbor RI/FS
Comprehensive Round 2 Report
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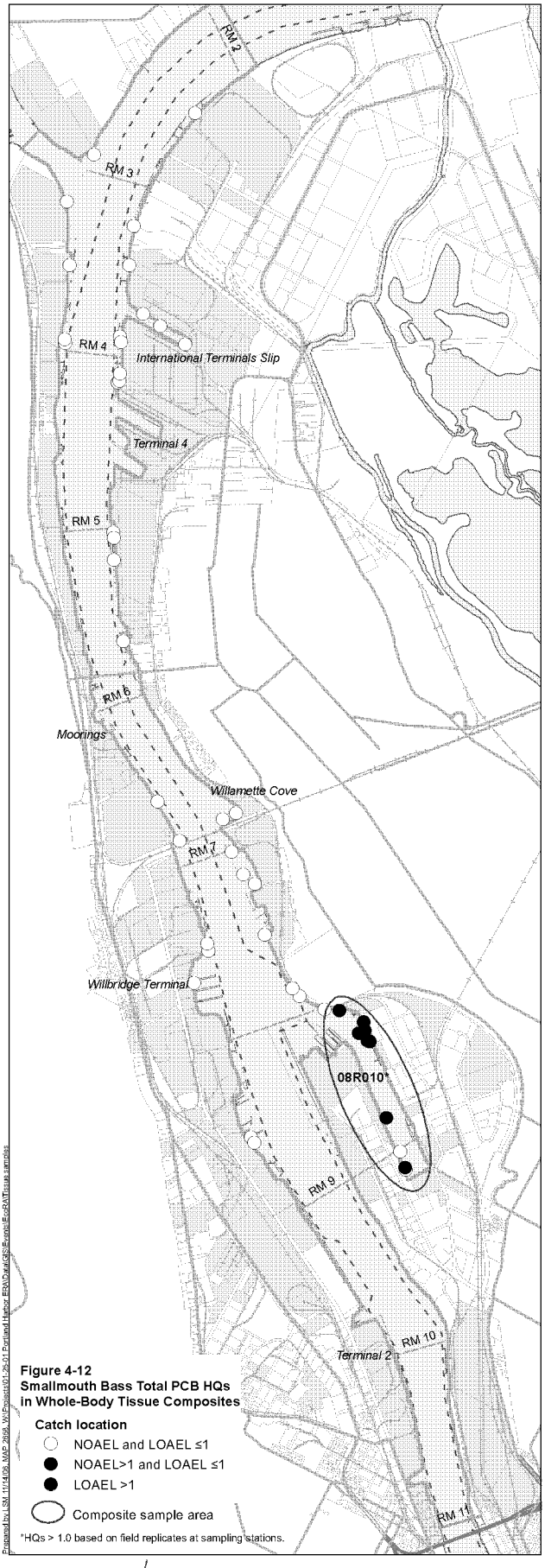

Figure 4-12
Smallmouth Bass Total PCB HQs in Whole-Body Tissue Composites

Catch location
- NOAEL and LOAEL ≤1
- NOAEL>1 and LOAEL ≤1
- LOAEL >1
- Composite sample area

*HQs > 1.0 based on field replicates at sampling stations.

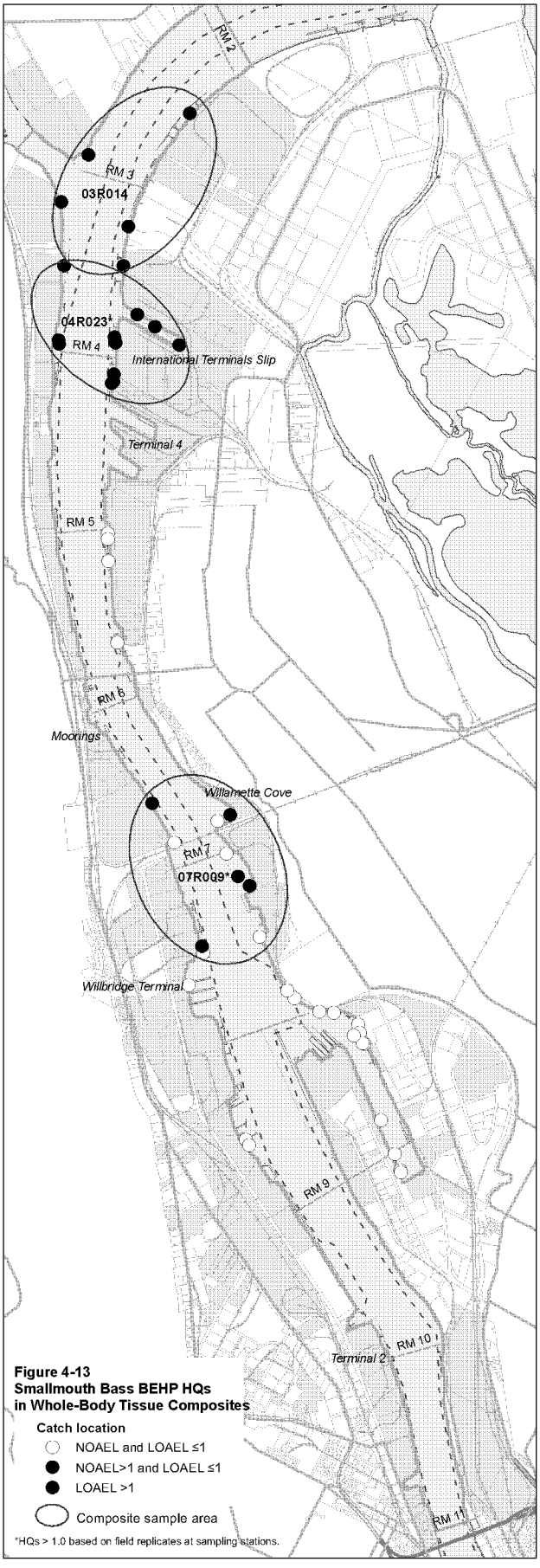

Figure 4-13
Smallmouth Bass BEHP HQs in Whole-Body Tissue Composites

Catch location
- NOAEL and LOAEL ≤1
- NOAEL>1 and LOAEL ≤1
- LOAEL >1
- Composite sample area

*HQs > 1.0 based on field replicates at sampling stations.

Prepared by LSM 11/14/06, MXP 2/9/8, W:\Projects\01-29-01 Portland Harbor ERA\Data\GIS\Bureau\EcoRAT\Tissue samples

Wind Ward LLC environmental

LWG Lower Willamette Group

N

0 1,000 2,000 3,000 4,000 Feet

Upland site
Waterfront tax lot
Freeway
Highway
Arterial
Railroad
Dock or structure
River Mile
Navigation Channel
13-foot contour (in-river analysis elevation)
River or slough

FEATURE SOURCES:
Transportation, Property, or Boundaries: Metro RLIS.
Channel & River miles: US Army Corps of Engineers.

Figure 4-12
Figure 4-13
River Miles 2 to 11

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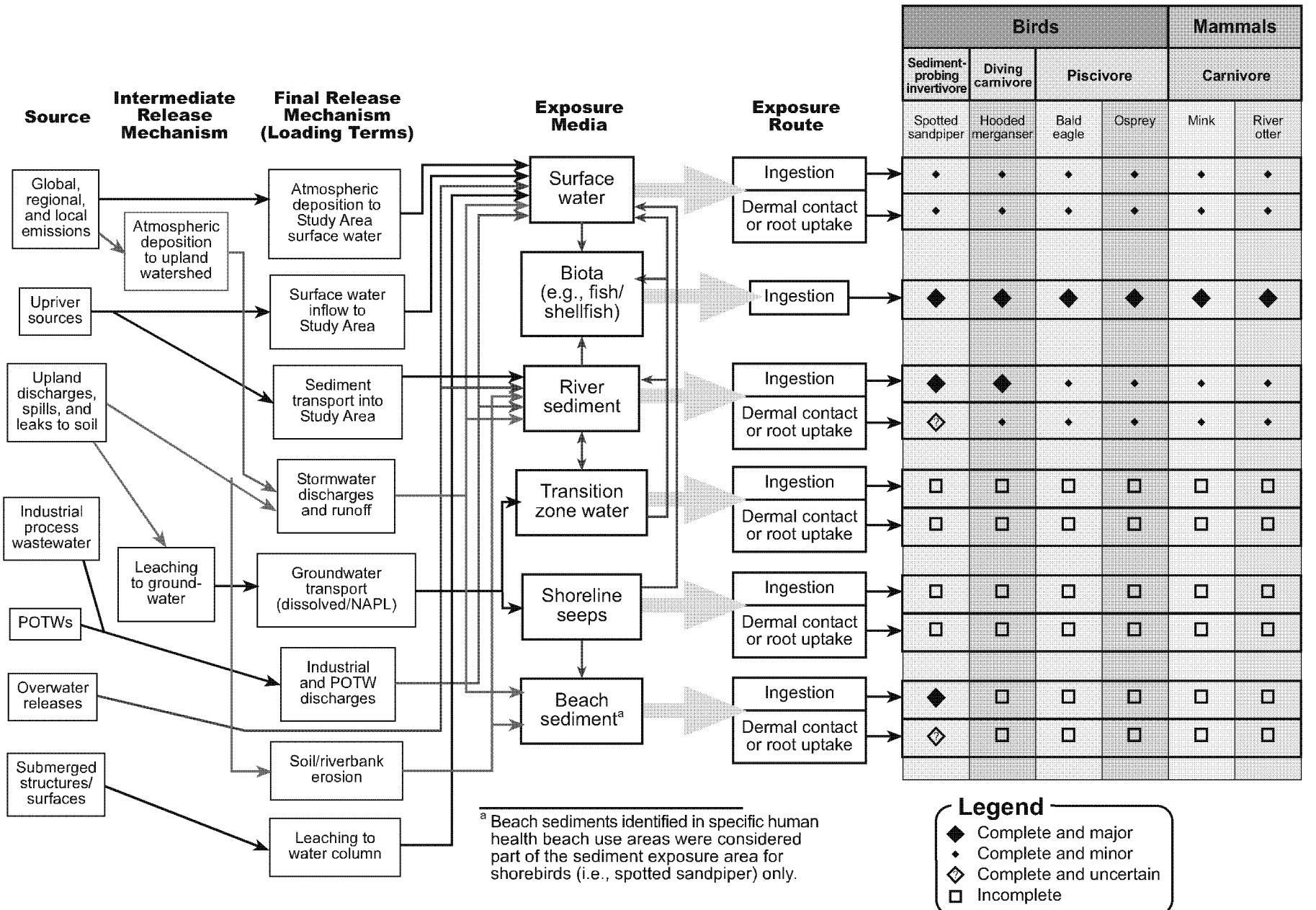

Figure 5-1
Preliminary Ecological Conceptual Site Model - Wildlife

Portland Harbor RI/FS
Comprehensive Round 2 Report
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Windward environmental LLC
LWG Lower Willamette Group

BZTO104(e)030715

Prepared by LSM 9/2/06. Modified by AWM 01/30/07. MAP 2505. W:\Projects\01-25-01 Portland Harbor FEA\Data\GIS\Events\EcoRAP\ecotoxrisk calcs\Shorebird

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Figure 5-2
Shorebird Beach Exposure Areas
River Miles 2 to 11

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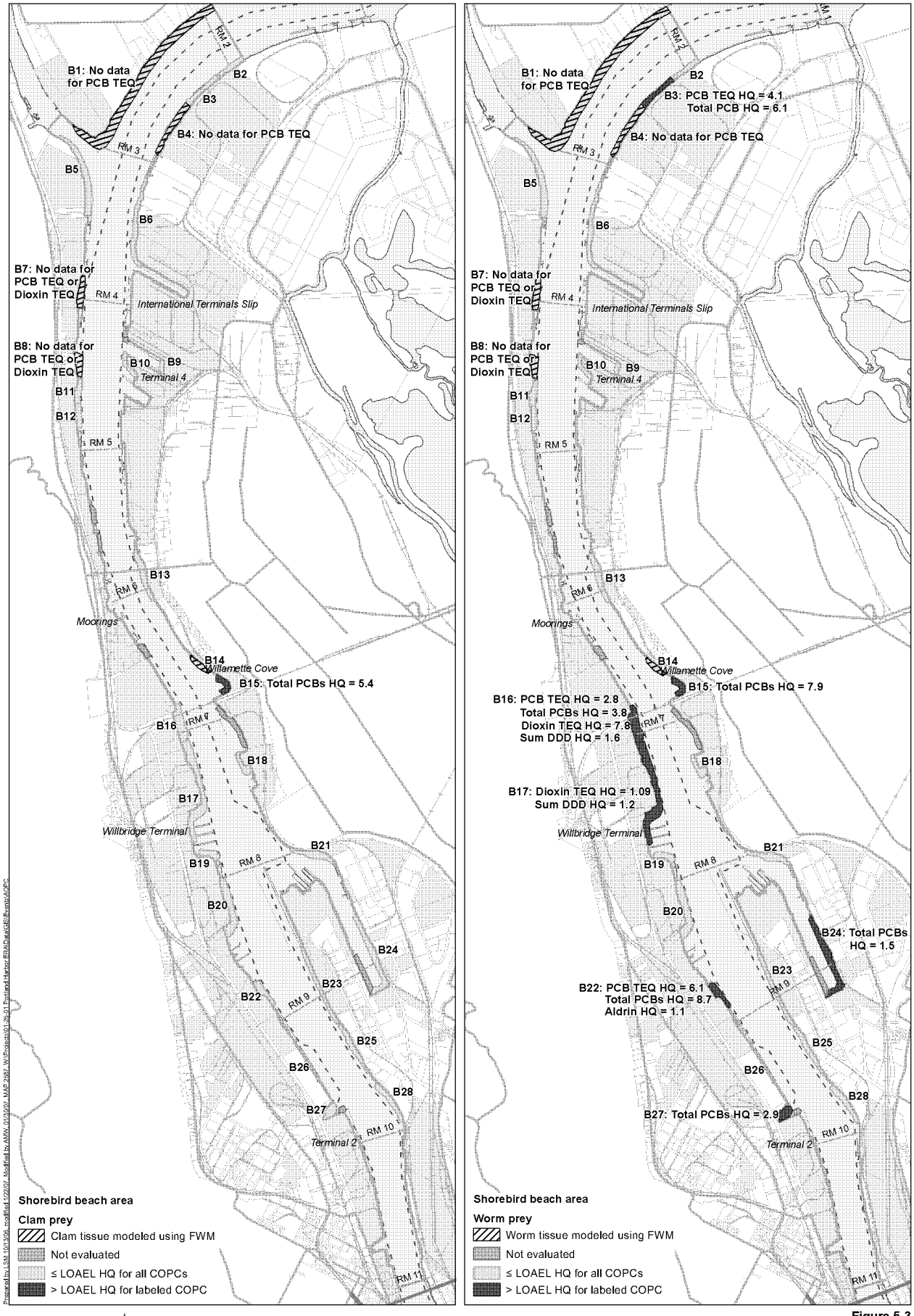

Prepared by LSM 10/13/06, modified 1/22/07. Modified by AMW, 01/30/07. MAP 2987. W:\Projects\01-25-01 Portland Harbor ERA\Data\GIS\Events\COPC

Wind Ward environmental LLC

LWG Lower Willamette Group

N

0 1,000 2,000 3,000 4,000 Feet

Upland site
Waterfront tax lot
Arterial
Highway
Freeway
Railroad
Dock or structure
River mile
Navigation channel
13-foot contour (in-river analysis elevation)
River or slough

FEATURE SOURCES:
Transportation, Property, or Boundaries: Metro RLIS.
Channel & River miles: US Army Corps of Engineers.

Figure 5-3
Shorebird Beach Areas Where LOAEL HQs >1 for Dioxin TEQ, PCB TEQ, Total PCBs, Aldrin, or Sum DDD Based on Clam or Worm Diet
River Miles 2 to 11

Portland Harbor RI/FS
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BZTO104(e)030717

Prepared by STS 10/24/06, Modified by AMW 01/30/07, Map 2697, W:\Projects\01-25-01 Portland Harbor ERA\Data\GIS\Exams\T7W

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Figure 6-1
Amphibian Exposure Areas and Surface Water Sampling Locations
River Miles 2 to 11

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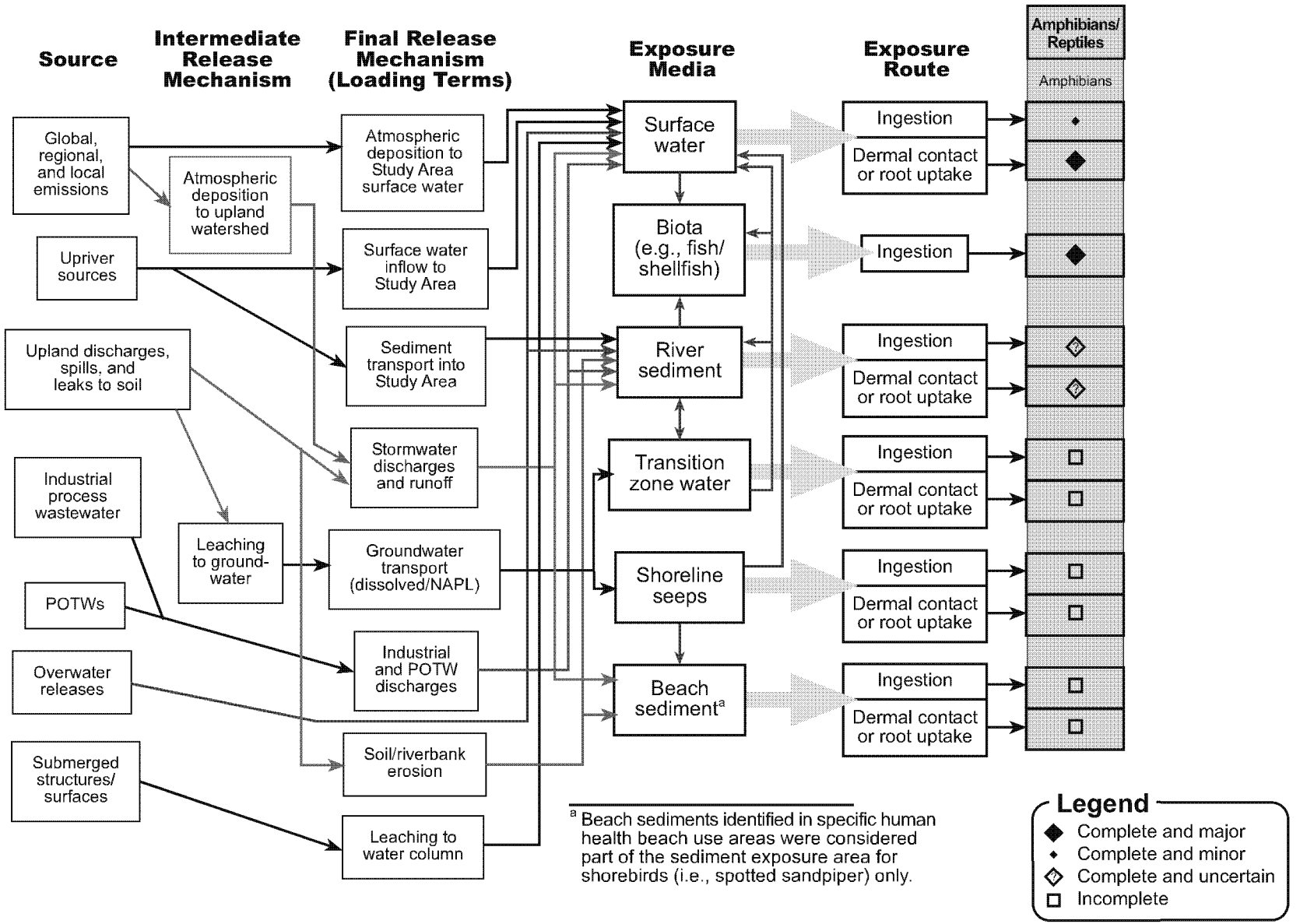

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Figure 6-2
Preliminary Ecological Conceptual Site Model - Amphibians

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BZTO104(e)030719

Prepared by STS 10/31/06, Modified by AMW 01/30/07, May 2004, W:\Projects\01-25-01 Portland Harbor ERA\Data\GIS\Exams\7.2\W

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Figure 7-1
Aquatic Plant Exposure Areas and Surface Water and Transition Zone Water Sampling Locations
River Miles 2 to 11

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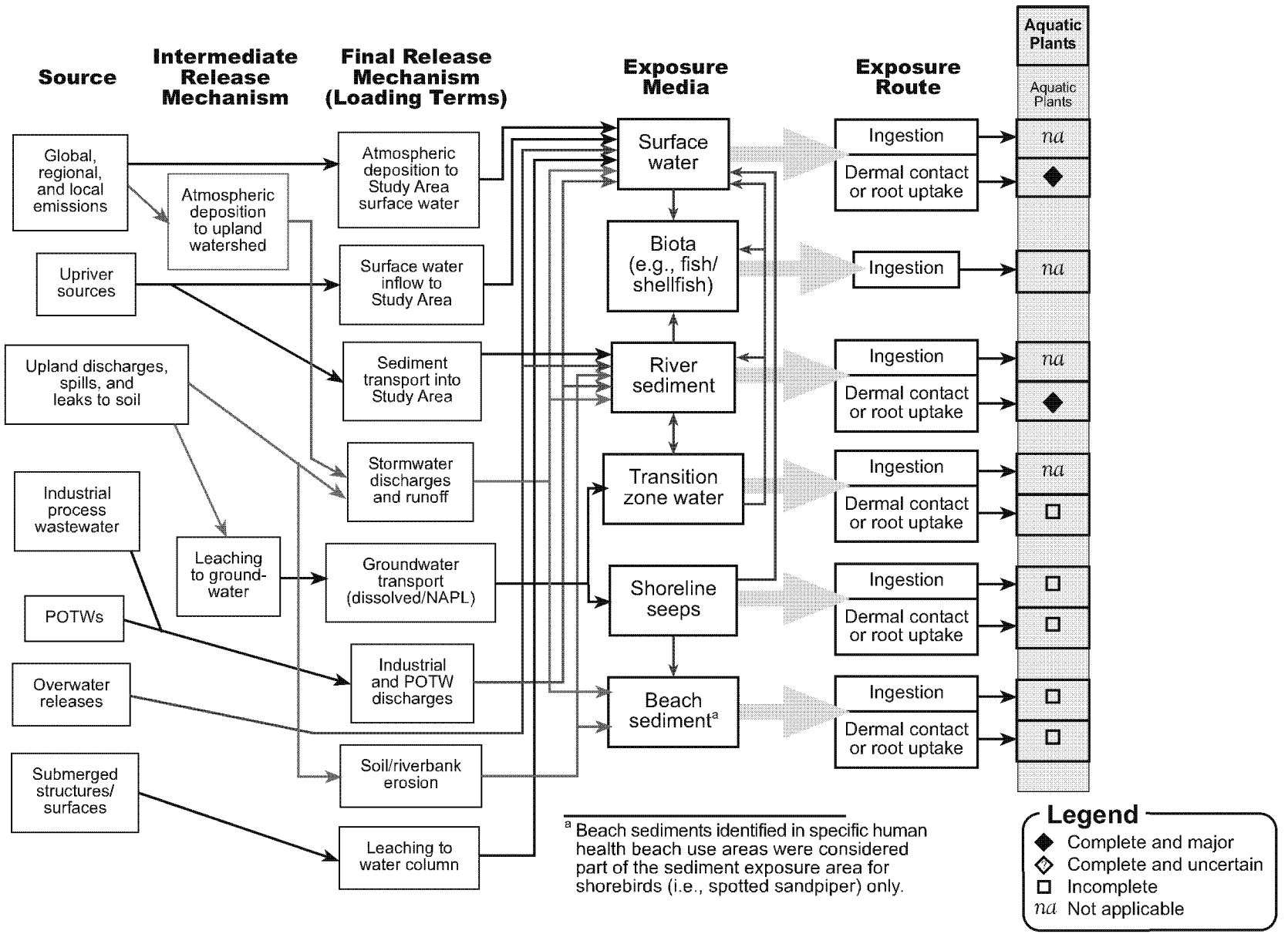

[a] Beach sediments identified in specific human health beach use areas were considered part of the sediment exposure area for shorebirds (i.e., spotted sandpiper) only.

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Figure 7-2
Preliminary Ecological Conceptual Site Model - Plants
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BZTO104(e)030721

Table 2-1. Summary of ERA Dataset

Medium	Data Types	Number of Samples
Surface sediment	all surface sediment collected within the 0-to-30.5-cm depth horizon in the Study Area	1,274
Sediment toxicity tests	surface sediment samples tested with *Chironomus tentans* and *Hyalella azteca*	233
Benthic invertebrate and whole-body fish tissue	field-collected clams, multiplate invertebrates, crayfish, black crappie, brown bullhead, carp, juvenile chinook salmon, largescale sucker, northern pikeminnow, peamouth, sculpin, smallmouth bass, and juvenile chinook salmon stomach contents; bioaccumulative lab worm and clam tissue	233
Surface water	surface water collected using a peristaltic pump and XAD-2 Infiltrex™ 300 system (column and filter) from Round 2 (events 1, 2, and 3)	125
Transition zone water (shallow)	all Round 2 shallow (0 to 38 cm) transition zone water sampled using a peeper (0 to 38 cm) and Trident® probe (30 cm)	196

XAD – XAD-2 Infiltrex™ 300 system

Table 2-2. Summary of Surface Sediment Sampling Events Included in the ERA Dataset

Sampling Event	Task Code	Sampling Dates	No. of Samples[a]	Chemicals Analyzed in Sediment
LWG-Collected Data				
Round 1 co-located surface sediment	Colocated1	October to November 2002	45	PCB Aroclors, butyltins, dioxins and furans, herbicides, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs, VOCs
Round 1 HHRA beach sediment[b]	HHRAbeach1	October 2002	22	PCB Aroclors, herbicides, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
Round 2A shorebird and HHRA beach sediment[b]	B01-01-48B_BK	July to November 2004	26	PCB Aroclors, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
Round 2A benthic sediment	B01-01-51B_BSE	December 2005	35	PCB Aroclors, butyltins, dioxins and furans, herbicides, metals, PAHs, PCB congeners, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs, VOCs
Round 2A groundwater pathway assessment co-located sediment grabs	B01-01-52B_SG	November and December 2005	38	butyltins, dioxins and furans, herbicides, metals, PAHs, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs, VOCs
Round 2A sediment cores[c]	B01-01-48B_SC	September to November 2004	47	PCB Aroclors, butyltins, dioxins and furans, metals, PAHs, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs
Round 2B sediment cores[c]	B01-01-48B_SCB	October 2005	34	PCB Aroclors, butyltins, dioxins and furans, herbicides, metals, PAHs, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs, VOCs
Round 2A sediment grabs[d]	B01-01-48B_SG	July to November 2004	528	PCB Aroclors, butyltins, dioxins and furans, herbicides, metals, PAHs, PCB congeners, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs, VOCs
Non-LWG-Collected Data				
2005 O&M dredge sediment characterization	WLCDRD05	May 2005	79	PCB Aroclors, butyltins, dioxins and furans, herbicides, metals, PAHs, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs, VOCs
City outfall pilot project	WLCOFH02	August 2002	18	PCB Aroclors, herbicides, metals, PAHs, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs
City outfall sediment investigation	WLCOFJ02	October 2002	85	herbicides, metals, PAHs, PCB congeners, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs

BZTO104(e)030723

Table 2-2. Summary of Surface Sediment Sampling Events Included in the ERA Dataset

Sampling Event	Task Code	Sampling Dates	No. of Samples[a]	Chemicals Analyzed in Sediment
Gasco source control evaluation	WLCGSD01	April 2001	5[c]	metals, PAHs, VOCs
McCormick & Baxter RI Phase 3	WLCMBJ99	October 1999	13	dioxins and furans, metals, PAHs, phenols
PAH in surface sediments	WLCASF97	June 1997	33	PAHs, SVOCs
Portland Harbor sediment investigation	WR-WSI98	September 1997	142	PCB Aroclors, butyltins, dioxins and furans, herbicides, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
Portland Shipyard environmental audit	PSYD&M97	November 1997 to December 1998	8	PCB Aroclors, butyltins, metals, PAHs, phthalates, SVOCs, VOCs
Portland Shipyard sediment investigation	PSYSEA98	March to April 1998	58	PCB Aroclors, butyltins, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs, VOCs
Terminal 4 EE/CA	WLCT4C04	March to May 2004	44[f]	PCB Aroclors, metals, PAHs, organochlorine pesticides, petroleum, phthalates, SVOCs
US Moorings sediment investigation 2002	WLCMRI02	September 2002	2	PCB Aroclors, butyltins, metals, PAHs, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs
Willamette River 1998 data	WRD&M98	January 1998	12	PCB Aroclors, butyltins, metals, PAHs, phthalates

[a] Includes field replicates.
[b] HHRA and shorebird beach samples were collected as transect composites of surface sediment along beach areas.
[c] Surface sediment samples from cores within the 0-to-30-cm depth were used in this ecological risk analysis.
[d] Benthic toxicity testing was conducted for 255 co-located samples from this sampling event. These data were used in the benthic risk assessment.
[e] Surface sediment from sediment cores were collected from both the 0-to-10- and 10-to-20-cm depth horizons at five locations. Data from the 0-to-10-cm depth were included in the ERA dataset; data from the 10-to-20-cm depth horizon was excluded from the ERA dataset.

EE/CA – engineering evaluation/cost analysis
HHRA – human health risk assessment
O&M – operation and maintenance
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RI – remedial investigation
SVOC – semivolatile organic compound
VOC – volatile organic compound

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Table 2-3. Summary of Benthic and Invertebrate Tissue Included in the ERA Dataset

Sampling Event and Dates	Species	No. of Samples[a]	Chemicals Analyzed in Tissue
Round 1A tissue sampling, June 2002	juvenile chinook salmon	6	PCB Aroclors, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
Round 1 tissue sampling, July to November 2002	black crappie[b]	4	PCB Aroclors, dioxins and furans, metals, PCB congeners, organochlorine pesticides, SVOCs
	brown bullhead[b]	6	PCB Aroclors, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	carp[c]	6	PCB Aroclors, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	clam	3	PCB Aroclors, butyltins, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
	crayfish	27	PCB Aroclors, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	largescale sucker	6	PCB Aroclors, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
	northern pikeminnow	6	PCB Aroclors, metals, organochlorine pesticides, SVOCs
	peamouth	4	PCB Aroclors, metals, organochlorine pesticides, SVOCs
	sculpin	27	PCB Aroclors, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	smallmouth bass	14	PCB Aroclors, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
Round 2A benthic tissue sampling, November to December 2005	clam	33	butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	bioaccumulative lab clam[d]	35	butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	bioaccumulative lab worm[d]	35	butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
Round 2A juvenile chinook salmon tissue sampling, May 2005	juvenile chinook salmon (stomach contents)[e]	5	PAHs, PCB congeners, organochlorine pesticides, SVOCs
	juvenile chinook salmon[e]	9	butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs

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Table 2-3. Summary of Benthic and Invertebrate Tissue Included in the ERA Dataset

Sampling Event and Dates	Species	No. of Samples[a]	Chemicals Analyzed in Tissue
Round 2A multiplate tissue sampling, September 2005	multiplate invertebrates	7	dioxins and furans, metals, PCB congeners, organochlorine pesticides, SVOCs

[a] Whole-body tissue composites only, except where noted; includes field replicates.
[b] Whole-body tissue was collected for black crappie and brown bullhead in support of the HHRA and was used to estimate dietary exposure for fish and wildlife receptors.
[c] Whole-body tissue was collected for carp in support of the HHRA and was used to estimate dietary exposure for fish and wildlife receptors. Carp was used as a surrogate ecological receptor only for dioxins and PCB congener data in whole-body tissue for the tissue-residue approach.
[d] Tissue chemistry analyses were based on bioaccumulative laboratory testing following exposure to field-collected Study Area surface sediment.
[c] Juvenile chinook salmon whole-body and stomach content tissue samples collected upstream of the Study Area near RM 17 were not included in the ecological risk analysis.
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound

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Table 2-4. Summary of Surface Water Data Included in the ERA Dataset

Sampling Event	Sampling Dates	Sampling Method	Number of Samples[a]	Chemicals Analyzed in Surface Water
Round 2A surface water event 1	November to December 2004	peristaltic pump	25	PCB Aroclors, butyltins, herbicides, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
		XAD (includes both filter and water column samples)	16	PCB Aroclors,[b] dioxins and furans, PAHs, PCB congeners, organochlorine pesticides, phthalates, SVOCs
Round 2A surface water event 2	March 2005	peristaltic pump	27	PCB Aroclors, butyltins, herbicides, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
		XAD (includes both filter and water column samples)	16	PCB Aroclors,[b] dioxins and furans, PAHs, PCB congeners, organochlorine pesticides, phthalates, SVOCs
Round 2A surface water event 3	July 2005	peristaltic pump	25	PCB Aroclors, butyltins, herbicides, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
		XAD (includes both filter and water column samples)	16	PCB Aroclors,[b] dioxins and furans, PAHs, PCB congeners, organochlorine pesticides, phthalates, SVOCs

[a] Includes field replicates.
[b] PCB Aroclors have been derived from individual PCB congeners by Axys Analytical Services, Ltd., for all the XAD samples for all three events.
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
XAD – XAD-2 Infiltrex™ 300 system

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Table 2-5. Summary of Transition Zone Water Data Included in the ERA Dataset

Sampling Site	Sampling Method	No. of Samples[a]	Chemicals Analyzed in Transition Zone Water
Arco Terminal 22T	peeper (unfiltered)	2	metals, PAHs, petroleum, SVOCs, VOCs
	Trident® (filtered)	7	metals, PAHs, petroleum, SVOCs
	Trident® (unfiltered)	7	metals, PAHs, petroleum, SVOCs, VOCs
Arkema – acid plant	peeper (unfiltered)	4	metals, PAHs, pesticides, SVOCs, VOCs
	Trident® (filtered)	4	metals, pesticides
	Trident® (unfiltered)	6	metals, PAHs, pesticides, SVOCs, VOCs
Arkema – chlorate plant	peeper (unfiltered)	4	metals, PAHs, SVOCs, VOCs
	Trident® (filtered)	6	metals
	Trident® (unfiltered)	6	metals, PAHs, SVOCs, VOCs
ExxonMobil Oil Terminal	Trident® (filtered)	12	metals, PAHs, petroleum, SVOCs
	Trident® (unfiltered)	11	metals, PAHs, petroleum, SVOCs, VOCs
Gasco	peeper (unfiltered)	3	metals, PAHs, petroleum, SVOCs, VOCs
	Trident® (filtered)	4	metals, PAHs, petroleum, SVOCs
	Trident® (unfiltered)	5	metals, PAHs, petroleum, SVOCs, VOCs
Gunderson	peeper (unfiltered)	6	metals, PAHs, SVOCs, VOCs
	Trident® (filtered)	2	metals
	Trident® (unfiltered)	3	metals, PAHs, SVOCs, VOCs
Kinder Morgan Linnton Terminal	peeper (unfiltered)	5	metals, PAHs, petroleum, SVOCs, VOCs
	Trident® (filtered)	3	metals, PAHs, petroleum, SVOCs
	Trident® (unfiltered)	4	metals, PAHs, petroleum, SVOCs, VOCs
Rhône-Poulenc	peeper (unfiltered)	2	herbicides, metals, PAHs, SVOCs, VOCs
	Trident® (filtered)	7	dioxins and furans, herbicides, metals, pesticides
	Trident® (unfiltered)	8	dioxins and furans, herbicides, metals, PAHs, pesticides, SVOCs, VOCs
Siltronic	peeper (unfiltered)	7	metals, PAHs, petroleum, SVOCs, VOCs
	Trident® (filtered)	6	metals, PAHs, petroleum, SVOCs
	Trident® (unfiltered)	6	metals, PAHs, petroleum, SVOCs, VOCs
	Geoprobe[b] (unfiltered)	41	metals, PAHs, SVOCs, VOCs
Willbridge Bulk Fuels Terminal	peeper (unfiltered)	3	metals, PAHs, petroleum, SVOCs, VOCs
	Trident® (filtered)	6	metals, PAHs, petroleum, SVOCs
	Trident® (unfiltered)	6	metals, PAHs, petroleum, SVOCs, VOCs

[a] Includes field replicates.

[b] Non-LWG collected TZW data from the Siltronic sampling site using a Geoprobe in May and June 2005 were included in the ERA dataset. All other TZW data were collected by LWG from October to December 2005.

PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound
VOC – volatile organic compound

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Table 3-1. Invertebrates Collected in the Study Area During Round 1 and Round 2

Phylum	Class/Subclass	Order	Family	Genus/Species	Common Name
Cnidaria	Hydrozoa	Hydroida	Hydridae	*Hydra* sp.	unknown
Platyhelminthes	Turbellaria			unknown	flatworm
Nematoda				unknown	roundworm
Nemertea	Enopla	Hoplonemertea	Tetrastemmatidae	*Prostoma* sp.	ribbon worm
				Prostoma graecense	ribbon worm
Mollusca	Gastropoda			unkown	snail
		Basommatophora	Physidae	*Physa* sp.	snail
			Ancylidae	*Ferrissia* sp.	limpet
			Planorbidae	*Menetus opercularis*	button sprite
	Bivalvia			unknown	clam
		Unionoida	Margaritiferidae	*Margaritifera falcata*	Western pearlshell mussel
		Unionoida	Unionidae	*Anodonta nuttalliana*	winged floater
		Veneroida	Corbiculidae	*Corbicula* sp.	Asiatic clam
			Sphaeriidea	*Pisidium* sp.	fingernail clam
Annelida	Polychaeta		Aeolosomatidae	unknown	worm
		Canalipalpata	Sabellidae	*Manayunkia speciosa*	sabellid worm
	Oligochaeta	Haplotaxida	Tubificidae	*Aulodrilus limnobius*	tubificid worm
				Aulodrilus sp.	tubificid worm
				Aulodrilu. pigueti	tubificid worm
				Aulodrilu. pluriseta	tubificid worm
				Branchiura sowerbyi	tubificid worm
				Limnodrilus hoffmeisteri	tubificid worm
				Quistradrilus multisetosus	tubificid worm
				unknown	tubificid worm
			Haplotaxidae	unknown	unknown
			Enchytraeidae	unknown	enchytraeid worm
			Naididae	*Chaetogaster* sp.	naidid worm
				Dero sp.	naidid worm
				Dero digitata	naidid worm
				Dero furcata	naidid worm
				Nais barbata	naidid worm
				Nais pardalis	naidid worm
				Nais variabilis	naidid worm
				Pristina aequiseta	naidid worm
				Pristina leidyi	naidid worm
				Pristina/Pristinella sp.	naidid worm
				Pristina osborni	naidid worm
				Slavina appendiculata	naidid worm
				Stylaria lacustris	naidid worm
				Vejdovskyella sp.	naidid worm
		Lumbriculidae		unknown	lumbricud worm
	Hirudinea			unknown	leech

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Table 3-1. Invertebrates Collected in the Study Area During Round 1 and Round 2

Phylum	Class/Subclass	Order	Family	Genus/Species	Common Name
		Euhirudinea	Erpobdellidae sp.	unknown	leech
Arthropoda	Malacostraca	Decapoda		unknown	crayfish
	Malacostraca	Isopoda	Asellidae	*Caecidotea* sp.	isopod
	Malacostraca	Amphipoda		unknown	amphipod
			Hyallelidae	*Hyallela* sp.	amphipod
			Corophiidae	*Corophium spinicorne*	amphipod
				Corophium sp.	amphipod
			Gammaridae	unknown	amphipod
				Anisogammarus sp.	amphipod
	Ostracoda			unknown	ostracod or seed shrimp
	Branchiopoda	Diplostraca	Sididae	*Sida crystallina*	daphnid
			Ilyocryptidae	*Ilyocryptus spinifer*	daphnid
				Ilyocryptus acutifrons	daphnid
	Maxillopoda	Calanoida	Diaptomidae	*Diaptomus* sp.	copepod
				unknown	copepod
	Insecta	Odonata	Gomphidae	unknown	dragonfly
				Stylurus sp.	dragonfly
		Diptera	Chironomidae	*Alabesmyia* sp.	midge
				Billia sp.	midge
				Bryophaenocladius sp.	midge
				Chironomini sp.	midge
				Chironomus sp.	midge
				Cladopelma sp.	midge
				Cladotanytarsus sp.	midge
				Corynoneura sp.	midge
				Cricotopus bicinctus gr.	midge
				Cricotopus sp.	midge
				Cryptochironomus sp.	midge
				Demeijerea sp.	midge
				Dicrotendipes sp.	midge
				Endochironomus sp.	midge
				Eukiefferiella brevicalcar gr.	midge
				Glyptotendipes sp.	midge
				Harnischia sp.	midge
				Nanocladius sp.	midge
				Orthocladius Complex	midge
				Parachironomus sp.	midge
				Paracladopelma sp.	midge
				Parakiefferiella sp.	midge
				Paralauterborniella sp.	midge
				Paralauterborniella nigrohalteris	midge
				Paraphaenocladius sp.	midge
				Paratanytarsus sp.	midge

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Table 3-1. Invertebrates Collected in the Study Area During Round 1 and Round 2

Phylum	Class/Subclass	Order	Family	Genus/Species	Common Name
				Phaenopsectra sp.	midge
				Polypedilum sp.	midge
				Procladius sp.	midge
				Psectrocladius sp.	midge
				Pseudochironomus sp.	midge
				Rheotanytarsus sp.	midge
				Stenochironomus sp.	midge
				Tanytarsus sp.	midge
				Thienemanniella sp.	midge
				Thienemanninyia gr. sp.	midge
				Xenochironomus xenolabis	midge
				unknown	midge
		Trichoptera sp.		unknown	caddisfly
			Hydroptilidae sp.	*Hydroptila* sp.	caddisfly
				Orthotrichia sp.	caddisfly
				unknown	caddisfly
			Leptoceridae	*Oecetis* sp.	caddisfly
			Polycentopodidae	*Polycentropus* sp.	caddisfly
				unknown	caddisfly
		Ephemeroptera	Caenidae	*Caenis* sp.	mayfly
			Heptageniidae	*Stenonema terminatum*	mayfly
	Arachnida	Acari		unknown	mite
		Acariformes	Oxidae	*Frontipoda* sp.	mite
			Hygrobatidae	*Hygrobates* sp.	mite
			Lebertiidae	*Lebertia* sp.	mite
			Limnesiidae	unknown	mite
			Torrenticolidae	*Torrenticola* sp.	mite
		Hydrachnida	Arrenuridae	*Arrenus* sp.	water mite
			Hygrobatidae	*Hygrobates* sp.	water mite
			Lebertiidae	*Lebertia* sp.	water mite
			Limnesiidae	*Limnesia* sp.	water mite
			Unionicolidae	*Unionicola* sp.	water mite
Bryozoa				unknown	“moss animals”

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Table 3-2. Lines of Evidence Used to Evaluate Risks to Benthic Invertebrate Receptors in the Study Area

Receptor of Concern	Assessment Endpoint	Measures of Exposure and Effect (Measurement Endpoints)	Lines of Evidence in Support of the Measurement Endpoints
Benthic community	survival, growth, and reproduction	sediment toxicity testing to empirically assess adverse effects	sediment toxicity testing (lethal and sublethal)
			concentrations in sediment compared to levels estimated by the empirically derived Portland Harbor predictive models to exhibit effects; models include endpoints for both species (*Hyalella azteca* and *Chironomus tentans*)
		bulk sediment concentrations vs. SQVs	consensus-based SQVs (TECs/PECs)
			empirical SQVs, PELs/TELs, LELs/SELs, SQS/CSLs, SQG-quotient, PEL-quotient, ERM-quotient
		benthic tissue data (modeled, laboratory, and field-collected) compared to tissue-based TRVs	empirical (Round 2 field-collected) whole-body concentrations relative to tissue TRVs
			empirical (Round 2 laboratory-exposed) whole-body concentrations relative to tissue TRVs
			predicted (BSAF) whole-body concentrations relative to tissue TRVs
		water exposure concentrations compared to AWQC or Eco SLs	concentrations in surface water relative to reported AWQC or Eco SLs
			concentrations in TZW relative to reported AWQC or Eco SLs[a]
			mechanistic-based SQVs derived from TZW; equilibrium partitioning
Shellfish (bivalves)	survival, growth, and reproduction	benthic tissue data (modeled, laboratory, and field-collected) compared to tissue-based TRVs	empirical (field-collected, Round 2) whole-body concentrations relative to tissue TRVs
			empirical (laboratory, Round 2) whole-body concentrations relative to tissue TRVs
			predicted (BSAF) whole-body concentrations relative to tissue TRVs; for TBT, a site-specific biota-sediment accumulation factor or a screening value based on sediment concentrations
		sediment toxicity testing to empirically assess adverse effects	*Hyalella* and *Chironomus* results used as bivalve surrogates
		water exposure concentrations compared to AWQC or Eco SLs	concentration in surface water relative to reported AWQC or ecological screening levels
			concentration in TZW relative to reported AWQC or Eco SLs
Crayfish	survival, growth, and reproduction	tissue data compared to tissue-based TRVs (chemical-dependent)	empirical whole-body concentrations relative to tissue TRVs
			predicted (BSAF) whole-body concentrations relative to tissue TRVs
		water exposure concentrations compared to AWQC or Eco SLs	concentrations in surface water relative to reported AWQC or Eco SLs
			concentrations in TZW relative to reported AWQC or Eco SLs

[a] Not included in Round 2 report owing to lack of data; consideration might be given to toxicity testing with TZW in Round 3, depending on the outcome of data gaps analysis.

AWQC – ambient water quality criteria
BSAF – biota-sediment accumulation factor
CSL – cleanup screening level
Eco SL – ecological screening level
ERM – effects range median
FWM – food web model
LEL – lowest effects level
PEL – probable effects level
SEL – severe effects level
SQG – sediment quality guidelines
SQS – sediment quality standards
SQV – sediment quality value
TBT – tributyltin
TEL – threshold effects level
TRV – toxicity reference value
TZW – transition zone water

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Table 3-3. Definitions of Biological Effects Levels

Test and Endpoint	Hit/No-Hit Criteria for Effects Levels[a]	
	Level 2 (80%)	**Level 3 (70%)**
Hyalella azteca 28-day survival	T/C < 0.8	T/C < 0.7
Hyalella azteca 28-day growth	T/C < 0.8	T/C < 0.7
Chironomus tentans 10-day survival	T/C < 0.8	T/C < 0.7
Chironomus tentans 10-day growth	T/C < 0.8	T/C < 0.7

[a] To be considered a toxic sediment at each of the two levels, the test response must also be statistically different from the negative control response (one-tailed test, $p < 0.05$).

T – mean of untransformed survival or weights in test sediment

C – mean of untransformed survival or weights in negative control sediment

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Table 3-4. Chemicals Screened Out Prior to Model Development

Chemicals with Fewer than 30 Detected Values[a]
1,2,3-Trichloropropane (1)
1,2,4-Trichlorobenzene (6)
1,2-Dichlorobenzene (5)
1,2-Dichloroethane (1)
1,4-Dichlorobenzene (5)
2,3,4,5-Tetrachlorophenol (5)
2,3,4,6/2,3,5,6-Tetrachlorophenol coelution (7)
2,4,5-Trichlorophenol (6)
2,4,6-Trichlorophenol (1)
2,4-D (6)
2,4-DB (1)
2,4-Dichlorophenol (2)
2,4-Dimethylphenol (1)
2-Chlorophenol (1)
2-Methylphenol (2)
4-Chloro-3-methylphenol (5)
4-Nitroaniline (1)
Acetone (4)
Aniline (8)
Benzene (19)
Benzyl alcohol (11)
Bis(2-chloroethyl)ether (2)
Carbon disulfide (12)
Chlorobenzene (13)
Chloroform (15)
Chromium, hexavalent (3)
Diethyl phthalate (7)
Dimethyl phthalate (12)
Di-n-octyl phthalate (5)
Endrin aldehyde (6)
Ethylbenzene (14)
Gasoline-range hydrocarbons (21)
Heptachlor (15)
Heptachlor epoxide (3)
Hexachlorobutadiene (21)
Hexachloroethane (26)
Isopropylbenzene (21)
m,p-Xylene (17)
MCPA (4-chloro-o-tolyloxyacetic acid) (2)
MCPP (2-(2-methyl-4-chlorophenoxy)propanoic acid) (1)
MTBE (methyl tertiary butyl ether)(6)

Table 3-4. Chemicals Screened Out Prior to Model Development

Chemicals with Fewer than 30 Detected Values[a]
Methyl tert-butyl ether (6)
Methyl ethyl ketone (20)
Mirex (4)
n-Nitrosodiphenylamine (2)
o-Xylene (23)
Styrene (1)
Toluene (5)
Trichloroethene (6)
Crustal Elements and Analytes Not Related to Toxicity
Aluminum (in the FPM)
Selenium (in the FPM)
Specific gravity
Total organic carbon (covaried with percent fines)
Total solids
Correlated Individual Chemicals Replaced by a Sum[b]
Individual grain size parameters (replaced by percent fines)
Individual dioxins and furans
Individual DDD, DDE, and DDT isomers and sums (replaced by total DDTs) – note that the last round of FPM used the 3 individual sums, whereas the LRM used just the total DDTs
Individual PAHs, LPAHs, HPAHs, dibenzofuran, and carbazole (replaced by total PAHs)
Individual Aroclors (replaced by total PCBs)
Individual endosulfans (replaced by total endosulfans in FPM; used individually in LRM)
Individual chlordanes, nonachlors, oxychlordane (replaced by total chlordane in FPM; used individually in LRM)

[a] Analytes were detected at least once; the number of times detected is shown in parentheses. Any analyte not listed in the table and not retained for model development was never detected.

[b] Sums used for FPM model development were consistent with sum definitions used throughout the Portland Harbor project; sums used in LRM development were calculated by NOAA, as described in Section 3.2.1

FPM – floating percentile model

HPAH – high-molecular-weight polycyclic aromatic hydrocarbon

LPAH – low-molecular-weight polycyclic aromatic hydrocarbon

LRM – logistic regression model

PAH – polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

Table 3-5. COIs Included in the Floating Percentile Model

Chemicals of Interest	
Metals	
Antimony	Lead
Arsenic	Mercury
Cadmium	Nickel
Chromium	Silver
Copper	Zinc
Butyltins	
Monobutyltin	Tributyltin
Dibutyltin	Tetrabutyltin
PAHs	
Total PAHs	
Phthalates	
BEHP	Dibutyl phthalate
Butylbenzyl phthalate	
SVOCs	
Hexachlorobenzene	
Phenols	
4-Methylphenol	Phenol
Pentachlorophenol	
PCBs	
Total PCBs	
Dioxins/Furans	
Total dioxins/furans	
Pesticides	
Aldrin	Endrin
Total chlordanes	Endrin ketone
Sum DDD	alpha-HCH
Sum DDE	beta-HCH
Sum DDT	gamma-HCH
Dieldrin	delta-HCH
Total endosulfans	Methoxychlor
TPH	
Diesel	Residual-range hydrocarbons
Conventionals	
Ammonia	Sulfides
Fines, percent (silt+clay)	

BEHP – bis(2-ethylhexyl) phthalate
HCH – hexachlorocyclohexane
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TPH – total petroleum hydrocarbons

Table 3-6. COIs Included in the Logistic Regression Model

Chemical	Number of Samples	Number of Detections	Chemical	Number of Samples	Number of Detections
Metals					
Aluminum	306	306	Iron	102	102
Antimony	279	188	Lead	598	595
Arsenic	461	460	Mercury	583	574
Barium	134	134	Nickel	483	470
Cadmium	598	583	Selenium	436	277
Chromium, total	502	501	Silver	518	503
Copper	545	541	Zinc	553	553
Butyltins					
Tributyltin	74	73			
PAHs					
2-Methylnaphthalene	538	380	Dibenzo(a,h)anthracene	617	456
Acenaphthene	615	430	Fluoranthene	645	583
Acenaphthylene	629	452	Fluorene	645	450
Anthracene	645	476	Indeno(1,2,3-c,d)pyrene	622	475
Benzo(a)anthracene	645	547	Naphthalene	631	425
Benzo(a)pyrene	645	539	Phenanthrene	645	554
Benzo(b)fluoranthene	612	525	Pyrene	645	586
Benzo(g,h,i)perylene	626	483	Total HPAHs	645	592
Benzo(k)fluoranthene	614	479	Total LPAHs	645	529
Carbazole	417	244	Total PAHs	660	607
Chrysene	645	567			
SVOCs					
Dibenzofuran	441	312	Hexachlorobenzene	427	195
Phthalates					
BEHP	437	290	Dibutyl phthalate	425	158
Butylbenzyl phthalate	437	131			
Phenols					
4-Methylphenol	418	150	Phenol	418	131
Pentachlorophenol	339	101			
PCBs					
Total PCBs	590	370			
Pesticides					
2,4′-DDD	302	238	Endosulfan sulfate	475	111
2,4′-DDE	302	127	Endrin	424	142
2,4′-DDT	302	163	Endrin ketone	471	172
4,4′-DDD	563	408	Heptachlor	510	126
4,4′-DDE	563	403	Heptachlor epoxide	543	110
4,4′-DDT	554	359	alpha-HCH	528	206

Table 3-6. COIs Included in the Logistic Regression Model

Chemical	Number of Samples	Number of Detections	Chemical	Number of Samples	Number of Detections
Total DDTs	342	301	beta-HCH	519	308
Aldrin	486	197	gamma-HCH	563	168
alpha-Chlordane-	519	263	delta-HCH	509	167
gamma-Chlordane	519	241	Methoxychlor	530	182
alpha-Endosulfan	511	124	cis-Nonachlor	302	102
beta-Endosulfan	486	143	trans-Nonachlor	301	114
Dieldrin	563	191	Toxaphene	544	105
TPH					
Diesel fuel	170	164	Residual-range organics	166	155
Oil and grease	104	104			
Conventionals					
Ammonia	273	273	Sulfides	267	237
Percent fines (silt and clay)	609	609			

BEHP – bis(2-ethylhexyl) phthalate
HCH – hexachlorocyclohexane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TPH – total petroleum hydrocarbons

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Table 3-7. Round 2 COPCs for Benthic Invertebrates

COPC	Field-Collected Benthic Invertebrates			Laboratory-Exposed Benthic Invertebrates		Predicted Tissue		
	Clam	Crayfish	Multiplate Invertebrate[a]	Clam	Worm	Field-Collected Clam	Laboratory-Exposed Clam	Worm
Metals								
Antimony								X
Arsenic					X			X
Cadmium	X				X	X		
Copper	X	X	X	X	X			
Zinc	X				X	X		
Butyltins								
TBT ion	X			X	X	X		X
PAHs								
Benzo(a)anthracene					X			
Benzo(a)pyrene					X			X
Benzo(b)fluoranthene					X			
Benzo(k)fluoranthene					X			
Chrysene					X			
Pyrene					X			X
Total PAHs	X			X	X	X		X
Phthalates								
BEHP				X				
Dibutyl phthalate	X				X			
PCBs								
Total PCBs	X				X			
Pesticides								
4,4′-DDD	X			X	X			
Total DDTs	X			X	X			

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Table 3-7. Round 2 COPCs for Benthic Invertebrates

COPC	Field-Collected Benthic Invertebrates			Laboratory-Exposed Benthic Invertebrates		Predicted Tissue		
	Clam	Crayfish	Multiplate Invertebrate[a]	Clam	Worm	Field-Collected Clam	Laboratory-Exposed Clam	Worm
beta-HCH								X
Endrin								X

[a] Invertebrates collected by multiplate samplers.

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TBT – tributyltin

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Table 3-8. Concentrations of Round 2 COPCs Identified Based on Field-Collected Clams

Round 2 COPC	Detection Frequency (%)	Unit (ww)	Minimum Detected Concentration	Maximum Detected Concentration	Mean Detected Concentration	UCL
Cadmium	100	mg/kg	0.05	0.218	0.094	0.11
Copper	100	mg/kg	6.85	13.5	9.6	10
Zinc	100	mg/kg	21.4	54	35	37
TBT ion	78	µg/kg	2.5	530	35	220
Total PAHs	100	µg/kg	34.8 J	4,980	618	891
Dibutyl phthalate	3	µg/kg	1,300	1,300	1,300	500
Total PCBs	100	µg/kg	50.1 J	2,660 J	245	576
4,4′-DDD	100	µg/kg	1.76	160 T	20	83
Total DDTs	100	µg/kg	7.82 J	463 J	57	130

COPC – chemical of potential concern
J – estimated concentration
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
T – value calculated or selected from multiple results
TBT – tributyltin
UCL – upper confidence limit

Table 3-9. Concentration of Round 2 COPC Identified Based on Field-Collected Crayfish

Round 2 COPC	Detection Frequency (%)	Minimum Detected Concentration (mg/kg)	Maximum Detected Concentration (mg/kg)	Mean Detected Concentration (mg/kg)	UCL
Copper	100	10.4	17.6	14	15

COPC – chemical of potential concern

Table 3-10. Concentration of Round 2 COPC Identified Based on Invertebrates Collected with Multiplate Samplers

Round 2 COPC	Detection Frequency (%)	Minimum Detected Concentration (mg/kg)	Maximum Detected Concentration (mg/kg)	Mean Detected Concentration (mg/kg)
Copper	100	3.01 J	6 J	4.5

COPC – chemical of potential concern
J – estimated concentration

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Table 3-11. Concentrations of Round 2 COPCs Identified Based on Laboratory-Exposed Clams

COPC	Detection Frequency (%)	Unit	Minimum Detected Concentration	Maximum Detected Concentration	Mean Detected Concentration	UCL
Copper	100	mg/kg	2.64 T	5.94 J	3.8	4.0
TBT ion	26	µg/kg	1.1	680	84	210
Total PAHs	100	µg/kg	19.2 J	1,320	139	600
BEHP	77	µg/kg	53 J	8,600	410	1,400
4,4′-DDD	100	µg/kg	0.375	702	24	220
Total DDTs	100	µg/kg	1.13 J	1,040	36.0	331

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
J – estimated concentration
PAH – polycyclic aromatic hydrocarbon
TBT – tributyltin

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Table 3-12. Concentrations of Round 2 COPCs Identified Based on Laboratory-Exposed Worms

COPC	Detection Frequency (%)	Unit	Minimum Detected Concentration	Maximum Detected Concentration	Mean Detected Concentration	UCL
Metals						
Arsenic	100	mg/kg	0.285	3.04	1.2	1.4
Cadmium	100	mg/kg	0.0364	0.254	0.069	0.083
Copper	100	mg/kg	1.83 T	20.2	2.9	3.8
Zinc	100	mg/kg	18.2 T	31.5	26	27
Butyltins						
TBT ion	43	µg/kg	2.1	1,700	120	540
PAHs						
Benzo(a)anthracene	100	µg/kg	5.9	2,600	200	1,000
Benzo(b)fluoranthene	100	µg/kg	7.7	1,600	150	760
Benzo(k)fluoranthene	100	µg/kg	3.9	1,500	110	610
Benzo(a)pyrene	100	µg/kg	2.2	1,500	100	610
Chrysene	100	µg/kg	14	3,900	350	1,700
Pyrene	100	µg/kg	16	11,000	870	4,600
Total PAHs	100	µg/kg	83.6	37,300	3,120	15,960
Phthalates						
Dibutyl phthalate	11	µg/kg	46	470 J	180	230
PCBs						
Total PCBs	100	µg/kg	44.8 J	4,310 J	612	2,390
Pesticides						
4,4′-DDD	100	µg/kg	4.08	1,060	71	420
Total DDTs	100	µg/kg	14.5 J	1,490	121	613

COPC – chemical of potential concern
J – estimated concentration
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TBT – tributyltin

Table 3-13. UCL Concentrations of Round 2 COPCs Based on the Predicted Tissue Line of Evidence

COPC	Unit	UCL[a]		
		Field-Collected Clam	Laboratory-Exposed Clam	Laboratory-Exposed Worm
Metals				
Antimony	mg/kg ww	NA	NA	0.0253
Arsenic	mg/kg ww	NA	NA	1.35
Cadmium	mg/kg ww	1.04	NA	NA
Copper	mg/kg ww	NA	3.97	NA
Zinc	mg/kg ww	35.7	NA	NA
Butyltins				
TBT ion	mg/kg lipid	2.51	8.81	7.23
PAHs				
Benzo(a)pyrene	mg/kg lipid	NA	NA	176
Pyrene	mg/kg lipid	NA	138	158
Total PAHs	mg/kg lipid	123	412	411
Pesticides				
beta-HCH	mg/kg lipid	NA	0.676	0.110
Endrin	mg/kg lipid	NA	5.59	1.97

[a] UCL is defined as the predicted 95UCL on the mean tissue concentration.
COPC – chemical of potential concern
HCH –hexachlorocyclohexane
LOE – line of evidence
NA – not available
PAH – polycyclic aromatic hydrocarbon
TBT – tributyltin
UCL – upper confidence limit
ww – wet weight

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Table 3-14. Metal Toxicity Studies for Invertebrates

Chemical	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Conditions	Endpoint	Source
Arsenic	grass shrimp (marine)	1.28	NA	water exposure for 28 days	growth	Lindsay and Sanders (1990)
Arsenic	grass shrimp (marine)	1.15	NA	water exposure for 28 days	growth	Lindsay and Sanders (1990)
Arsenic	snail (freshwater)	3.6 – 16	NA	water exposure for 28 days	survival	Spehar et al. (1980)
Cadmium	mysid shrimp (marine)	NA	1.29	water exposure for 33 days	growth	Carr et al. (1985)
Cadmium	red swamp crayfish (freshwater)	0.5	NA	single injection assessed after 21 days	reproduction	Reddy et al. (1997)
Cadmium	grass shrimp (marine)	0.6	NA	sediment exposure for 14 days	survival	Rule and Alden (1996)
Cadmium	grass shrimp (marine)	NA	2.6	water exposure for 21 days	survival	Vernberg et al. (1977)
Cadmium	crayfish	14.9	22	water exposure for 5 months	survival	Thorp et al. (1979)
Cadmium	crayfish (freshwater)	NA	28.4	water exposure for 2 weeks	survival	Mirenda (1986b)
Cadmium	crayfish (freshwater)	534	NA	water exposure for approx. 8 days	survival	Gillespie et al. (1977)
Cadmium	hard clam (marine)	0.46	NA	sediment exposure for 14 days	survival	Rule et al. (1996)
Cadmium	eastern elliptio	18	NA	water exposure for 60 days	survival	Tessier et al. (1996)
Cadmium	snail (freshwater)	9	30	water exposure for 28 days	survival	Spehar et al. (1978)
Cadmium	banded mystery snail	34	NA	water exposure for 60 days	survival	Tessier et al. (1996)
Cadmium	Eastern oyster (marine)	18.2	54	water exposure for 37 weeks	reproduction	Zaroogian and Morrison (1981)
Cadmium	Eastern oyster (marine)	58.4	NA	water exposure for 40 weeks	survival	Zaroogian (1980)
Cadmium	snail	460	625	sediment exposure for 10 days	survival	Carlson et al. (1991)
Copper	grass shrimp (marine)	40	NA	sediment exposure for 14 days	survival	Rule et al. (1996)
Copper	crayfish (freshwater)	50	NA	water exposure for 48 hours	survival	Evans (1980)
Copper	hard clam (marine)	6.4	NA	sediment exposure for 14 days	survival	Rule et al. (1996)
Copper	Pacific littleneck clam (marine)	NA	9.34	water exposure for 30 days	survival	Roesijadi (1980)
Copper	Baltic macoma (marine clam)	NA	12	Combination of water and diet exposure for 17 days	survival	Absil et al. (1996)
Copper	clam (marine)	NA	201	water exposure for up to 184 hours	survival	Kumaraguru et al. (1980)
Zinc	crayfish (freshwater)	12.7	35.2	water exposure for 2 weeks	survival	Mirenda (1986a)

[a] Dry weight concentrations were converted to wet weight, assuming 80% moisture in the organism (wet weight concentration = 0.2 X dry weight concentration)

LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
NA – not available
ww – wet weight

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Table 3-15. TBT Toxicity Studies with Invertebrates

Chemical	Test Species	NOAEL (µg/kg dw)	LOAEL (µg/kg dw)	Exposure Conditions	Endpoint	Source
Tributyltin ion	polychaete (*Armandia brevis*) (marine)	970	2,360	sediment exposure for 42 days	reduced growth	Meador and Rice (2001)
Tributyltin ion	polychaete (*Neanthes arenaceodentata*) (marine)	2,990	6,270	water exposure for 10 weeks	impaired reproduction	Moore et al. (1991)
Tributyltin ion	blue mussel (*Mytilus edulis)* (marine)	3,960	5,440	water exposure for 4 days	reduced growth	Widdows and Page (1993)
Tributyltin ion	amphipod (*Hyalella azteca*)	NA	32,000	water exposure for 10 days	reduced survival (LC50)	Borgmann et al. (1996)
Tributyltin ion	polychaete (*Armandia brevis*) (marine)	NA	41,000	water exposure for 10 days	reduced survival (LC50)	Meador (1997)
Tributyltin ion	amphipod (*Eohaustorius estuarius*) (marine)	NA	59,000	water exposure for 10 days	reduced survival (LC50)	Meador (1997)
Tributyltin ion	amphipod (*Rhepoxynius abronius)* (marine)	NA	54,000	water exposure for 10 days	reduced survival (LC50)	Meador (1997)

[a] Dry weight concentrations were converted to wet weight, assuming 80% moisture in the organism (wet weight concentration = 0.2 X dry weight concentration)

dw – dry weight
LC50 – concentration that is lethal to 50% of an exposed population
LOAEL – lowest-observed-adverse-effect level
NA – not available
NOAEL – no-observed-adverse-effect level
TBT – tributyltin

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Table 3-16. PAHs and Phthalates Toxicity Studies with Invertebrates

Chemical	Test Species	NOAEL (µg/kg ww)	LOAEL (µg/kg ww)	Exposure Conditions	Endpoint	Source
Fluoranthene	blue mussel (marine)	NA	222	water exposure for 4 weeks	reproduction	Eertman et al. (1995)
Benzo(a)-pyrene	blue mussel (marine)	NA	302	water exposure for 4 weeks	reproduction	Eertman et al. (1995)
Benzo(a)-pyrene	fingernail clam (marine)	1,250	NA	sediment exposure for 5 days	survival	Borchert et al. (1997)
PAHs	zebra mussel (freshwater)	6,580 (5,600 – 10,400)	NA	water and sediment exposure for 34 days	survival	Roper et al. (1997)
Di(2-ethylhexyl) phthalate	penaied shrimp (marine)	18,300	NA	diet exposure for 14 days	survival	Hobson et al (1984)
Phthalates	crustaceans (freshwater)	320 – 26,800	NA	water exposure for 21 days	reproduction	Brown and Thompson (1982), as cited in Dillon (1984)

[a] Dry weight concentrations were converted to wet weight, assuming 80% moisture in the organism (wet weight concentration = 0.2 X dry weight concentration).

NA – not available

PAH – polycyclic aromatic hydrocarbon

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Table 3-17. PCB Toxicity Studies with Invertebrates

Chemical	Test Species	NOAEL (µg/kg ww)	LOAEL (µg/kg ww)	Exposure Conditions	Endpoint	Source
PCBs (Aroclor 1254)	crayfish (freshwater)	1,220	NA	water exposure for 21 days	survival	Sanders and Chandler (1972)
PCBs (Aroclor 1254)	pink shrimp (marine)	1,300	3,900	water exposure for 48 hours	survival	Duke et al. (1970)
PCBs (Aroclor 1254)	pink shrimp (marine)	NA	16,000	water exposure for 20 days	survival	Duke et al. (1970)
PCBs (Aroclor 1254)	blue crab (marine)	23,000	NA	water exposure for 20 days	survival	Duke et al. (1970)
PCBs (Aroclor 1254)	grass shrimp (marine)	18,000	27,000	water exposure for 16 days	survival	Nimmo et al. (1974)
PCBs (Aroclor 1016)	grass shrimp (marine)	NA	1,100	water exposure for 96 hours	survival	Hansen et al. (1974)
PCBs (Aroclor 1016)	brown shrimp (marine)	3,800	42,000	water exposure for 96 hours	survival	Hansen et al. (1974)
PCBs (Aroclor mixture)	Asiatic clam freshwater)	900	NA	water exposure for 4 weeks	survival	Peterson et al. (1994)
PCBs (mixture of 13 congeners)	bent nose clam (marine)	1,750[a]	NA	sediment exposure for 119 days	survival/growth	Boese et al. (1995)
Aroclor 1016	American oyster (marine)	NA	4,000	water exposure for 96 hours	growth (shell growth)	Hansen et al. (1974)
Aroclor 1254	oyster (*Crassostrea virginica*) (marine)	8,100	33,000	water exposure for 96 hours	growth (shell growth rate)	Duke et al. (1970)
Aroclor 1254	young oysters (*Crassostrea virginica*) (marine)	101,000	119,000	water exposure for 24 weeks	growth (height and body weight after 6 weeks)	Lowe et al. (1972)

[a] Dry weight concentrations were converted to wet weight, assuming 80% moisture in the organism (wet weight concentration = 0.2 X dry weight concentration)

NA – not available

PCB – polychlorinated biphenyl

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Table 3-18. DDT and Endrin Toxicity Studies with Invertebrates

Chemical	Test Species	NOAEL (µg/kg ww)	LOAEL (µg/kg ww)	Exposure Conditions	Endpoint	Source
4,4′-DDD	*Hyalella azteca* (amphipod)	NA	15,000	water exposure for 10 days	survival	Lotufo et al. (2000)
4,4′-DDD	*Diporeia* spp. (amphipod)	NA	116,000	water exposure for 10 days	survival	Lotufo et al. (2000)
4,4′-DDD	*Diporeia* spp. (amphipod)	NA	84,000	water exposure for 28 days	survival	Lotufo et al. (2000)
DDTs	crayfish (adult freshwater)	46	NA	water exposure for 3 days	survival	Johnson et al. (1971)
DDTs	pink shrimp (marine)	NA	60	water exposure for 56 days	survival	Nimmo et al. (1970)
DDTs	blue crab (juvenile marine)	26	200	treated food for 5 weeks	metabolic rate	Leffler (1975)
4,4′-DDT	*Hyalella azteca* (amphipod)	NA	2,100	water exposure for 10 days	survival	Lotufo et al. (2000)
4,4′-DDT	*Diporeia* spp. (amphipod)	NA	30,000	water exposure for 10 days	survival	Lotufo et al. (2000)
4,4′-DDT	*Diporeia* spp. (amphipod)	NA	16,000	water exposure for 28 days	survival	Lotufo et al. (2000)
DDT (4,4′-DDT)	zebra mussel (freshwater)	16 – 21	27	field exposed	reproduction (histo-pathological)	Binelli et al. (2001)
Endrin	grass shrimp (marine)	48	80	water exposure for two generations	growth	Tyler-Schroeder (1979)
Endrin	grass shrimp (marine)	80	176	water exposure for 4.5 months	survival	Tyler-Schroeder (1979)
Endrin		260	16,400	water exposure for 1 week	survival	Mason and Rowe (1976)

[a] Dry weight concentrations were converted to wet weight, assuming 80% moisture in the organism (wet weight concentration = 0.2 X dry weight concentration).

LOAEL – lowest-observed-adverse-effect level

NA – not available

NOAEL – no-observed-adverse-effect level

ww – wet weight

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Table 3-19. Round 2 COPCs for Benthic Invertebrates Based on Near-Bottom Surface Water Samples

Round 2 COPC	Sampling Method	
	XAD	Peristaltic Pump
Zinc (dissolved)		X
Benzo(a)anthracene		X
Benzo(a)pyrene		X
4-Chloro-3-methylphenol		X
Total PCBs		X
2,4′-DDD	X	
2,4′-DDT		X
4,4′-DDD	X	
4,4′-DDT	X	X
Total DDTs	X	X

COPC – chemical of potential concern
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
XAD – Infiltrex 300 system with XAD-2 resin column

Table 3-20. Near-Bottom Surface Water Round 2 COPCs

Chemical	Detection Frequency	Unit	Minimum Detect	Maximum Detect	Mean Detect	UCL
Zinc (dissolved)	49/56	mg/L	0.0009	0.0419	0.0029	0.0058
Benzo(a)anthracene	25/71	µg/L	0.000118 J	0.11	0.0079	0.013
Benzo(a)pyrene	21/71	µg/L	0.0000183 J	0.15	0.010	0.016
4-Chloro-3-methylphenol	6/55	µg/L	0.068 J	0.65	0.21	0.12
Total PCBs	20/53	µg/L	0.000379 J	0.0162 J	0.0046	0.00428
2,4′-DDD	15/58	µg/L	0.00000566 J	0.002057	0.00041	0.0021
2,4′-DDT	15/58	µg/L	0.00000103 J	0.0187 NJ	0.0013	0.0010
4,4′-DDD	16/58	µg/L	0.00001201	0.00325	0.00084	0.0040
4,4′-DDT	17/58	µg/L	0.000001342 J	0.003862	0.000596	0.00591
Total DDTs	19/58	µg/L	0.0000496 J	0.0199 NJ	0.0030	0.0102

J – estimated concentration
N – presumptive evidence of a compound
T – value calculated or selected from multiple results

Table 3-21. Chemical Concentrations for Round 2 COPCs (Zinc, PAHs, Phenol, and PCBs) in Near-Bottom Surface Water

Location	Event[a]	Sampling Method	Zinc (dissolved) (mg/L)	Benzo(a) anthracene (µg/L)	Benzo(a) pyrene (µg/L)	4-Chloro-3-methylphenol (µg/L)	Total PCBs (µg/L)[b]
W001	1	peristaltic	0.0014	0.0084 U	0.0064 U	0.029 U	0.0025 UJ
	2		0.0014 T	0.0042 U	0.0032 U	0.032 U	0.0025 U
	3		0.00121 UJ	0.0021 U	0.0016 U	0.048 U	0.008 J
W002[c]	1	peristaltic	0.0023	0.0021 U	0.0016 U	0.029 U	NA
	2		0.0012	0.0042 U	0.0032 U	0.41 U	0.00255 U
	2 (FR)		0.0011	0.0042 U	0.0032 U	0.68 U	NA
	3		0.00117 UJ	0.0021 U	0.0016 U	0.032 U	0.0025 UJ
	3 (FR)		0.00146 UJ	0.0021 U	0.0016 U	0.062 U	NA
W003	1	peristaltic	0.0009	0.0084 U	0.0064 U	0.091 J	0.0025 UJ
	2		0.0018	0.0042 U	0.0032 U	0.65	0.0025 U
	3		0.0019 J	0.003 J	0.0027 J	0.03 U	0.0025 UJ
W004	1	peristaltic	0.0025	0.012 J	0.0064 U	0.029 U	0.00842 J
	2		0.0043	0.0047 U	0.0049 U	0.03 U	0.0162 J
	2		NA	NA	NA	NA	0.0135
	3		0.00188 J	0.0021 U	0.0016 U	0.074 U	0.0025 UJ
W006	1	peristaltic	0.0028	0.0059 J	0.0032 UJ	0.068 J	0.00256 U
	2		0.0018	0.0042 U	0.0032 U	0.53 U	0.00252 U
	3		0.002 J	0.0042 U	0.0032 U	0.031 U	0.0025 U
W007	1	peristaltic	0.0015	0.0042 UJ	0.0032 UJ	0.082 J	0.0026 U
	2		0.0017	0.0042 U	0.0032 U	0.031 U	0.00261 U
	3		0.0024 J	0.0065 J	0.004 J	0.031 U	0.0025 U
W008	1	peristaltic	0.0019	0.0046 J	0.0033 J	0.13 U	0.0025 U
	2		0.0042	0.0042 U	0.0032 U	0.031 U	0.0025 U
	3		0.0047 J	0.0078 J	0.0075 J	0.032 U	0.0025 U
W009	1	peristaltic	0.0017	0.0043 J	0.0032 U	0.07 U	0.0025 U
	2		0.0016	0.0042 U	0.0032 U	0.03 U	0.0025 U
	3		0.0018 J	0.0042 U	0.0032 U	0.03 U	0.0025 U
W012	1	peristaltic	0.0015	0.019	0.02	0.051 U	0.0025 UJ

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Table 3-21. Chemical Concentrations for Round 2 COPCs (Zinc, PAHs, Phenol, and PCBs) in Near-Bottom Surface Water

Location	Event[a]	Sampling Method	Zinc (dissolved) (mg/L)	Benzo(a) anthracene (µg/L)	Benzo(a) pyrene (µg/L)	4-Chloro-3-methylphenol (µg/L)	Total PCBs (µg/L)[b]
	2		0.0017	0.006 U	0.0082 U	0.031 U	0.0026 U
	3		0.0018 J	0.11	0.15	0.15 U	0.0025 U
W013-1	1	peristaltic	0.0009 J	0.0084 U	0.0064 U	0.029 U	NA
	1	XAD	NA	0.000542 J	0.0004219 J	NA	0.00334 J
	2	peristaltic	0.0014	0.0021 U	0.0016 U	0.074 J	NA
	2	XAD	NA	0.000571 J	0.0000183 J	NA	0.0120 J
	3	peristaltic	0.00163 UJ	0.0021 U	0.0016 U	0.033 U	NA
	3	XAD	NA	0.000996	0.000843 J	NA	0.00798 J
W013-2 (FR)[d]	1	peristaltic	0.0022 U	0.0084 U	0.0064 U	0.029 U	NA
	1	XAD	NA	0.0004295 J	0.000336	NA	0.00251 J
	2	peristaltic	0.0018	0.0028 J	0.0024 J	NA	NA
	2	XAD	NA	0.000118 J	0.0000316 J	NA	0.00206 J
	3		NA	0.000544	0.000232 U	NA	0.00216
W015	1	peristaltic	0.0022	0.0042 U	0.0032 U	0.12 U	NA
	2		0.0018	0.0042 U	0.0032 U	0.031 U	NA
	3		0.0016 J	0.0042 U	0.0032 U	0.14 U	NA
	1	XAD	NA	0.003352	0.003033 J	NA	0.00193 J
	2		NA	0.001601	0.0017532 J	NA	0.000541 J
	3		NA	0.00613	0.006636	NA	0.000807 J
W016-1	1	peristaltic	0.0014 U	0.0042 U	0.0032 U	0.088 U	NA
	2		0.0017	0.0042 U	0.0032 U	0.03 U	NA
	3		0.00215 J	0.0042 U	0.0032 U	0.067 U	NA
	1	XAD	NA	0.000856 J	0.0008887 J	NA	0.000379 J
	2		NA	0.000643	0.0007898 J	NA	0.00044 9 J
	3		NA	0.00374	0.00257	NA	0.00128 J
W016-2 (FR)	1	peristaltic	0.0016 U	0.0042 UJ	0.0032 UJ	0.031 UJ	NA
W017	1	peristaltic	0.0016	0.0021 U	0.0016 U	0.044 U	NA
	2		0.0017	0.0042 U	0.0032 U	0.031 U	0.00256 U

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Table 3-21. Chemical Concentrations for Round 2 COPCs (Zinc, PAHs, Phenol, and PCBs) in Near-Bottom Surface Water

Location	Event[a]	Sampling Method	Zinc (dissolved) (mg/L)	Benzo(a) anthracene (µg/L)	Benzo(a) pyrene (µg/L)	4-Chloro-3-methylphenol (µg/L)	Total PCBs (µg/L)[b]
	3		0.0018 J	0.0042 U	0.0032 U	0.063 U	0.0025 U
W018[c]	1	peristaltic	0.0049	0.0084 U	0.0064 U	0.029 U	NA
	1	XAD	NA	0.0003815 J	0.000384	NA	0.000439 J
	2	peristaltic	0.0023	0.0021	0.0016 U	0.32 J	NA
	2	XAD	NA	0.001574	0.00161	NA	0.00169 J
	3	peristaltic	0.00267	0.0042 U	0.0032 U	0.031 U	NA
	3	XAD	NA	0.000167	0.0000455 J	NA	0.000830 J
W019	1	peristaltic	0.0016	0.0042 U	0.0032 U	0.041 U	0.00253 U
	2		0.0017	0.0042 U	0.0032 U	0.031 U	0.00263 U
	3		0.00174	0.0042 U	0.0032 U	0.031 U	0.0025 U
W021	1	peristaltic	0.0032	0.0042 U	0.0032 U	0.034 U	0.0025 U
	2		0.002	0.0042 U	0.0032 U	0.032 U	0.0026 U
	3		0.00161	0.0042 U	0.0032 U	0.031 U	0.0025 U
W022	1	peristaltic	0.0419	0.0042 U	0.0032 U	0.046 U	0.0025 U
	2		0.0037	0.0042 U	0.0032 U	0.029 U	0.0025 U
	3		0.0027	0.0042 U	0.0032 U	0.03 U	0.00717 J

[a] Sampling Event 1 – November 2004; Sampling Event 2 – March 2005; Sampling Event 3 – July 2005.
[b] Total PCBs were calculated as total PCBs Aroclors for samples collected with the peristaltic pump
[c] PCB Aroclors were not analyzed at W002 during the November 2004 sampling event.
[d] There is no peristaltic data (only XAD data) for W013-2, Event 3.
[e] PCB Aroclors and pesticides were not analyzed at W018 during the March and July 2005 sampling events using the peristaltic method.

FR – field replicate
J – estimated concentration
N – presumptive evidence of a compound
NA – not analyzed
T – value calculated or selected from multiple results
U – not detected at given concentration

Table 3-22. Chemical Concentrations for Round 2 COPCs (Pesticides) in Near-Bottom Surface Water

Location	Event[a]	Sampling Method	2,4′-DDD (µg/L)	2,4′-DDT (µg/L)	4,4′-DDD (µg/L)	4,4′-DDT (µg/L)	Total DDTs (µg/L)
W001	1	peristaltic	0.000472 UJ	0.000472 UJ	0.000472 UJ	0.000472 UJ	0.000472 UJ
	2		0.00049 U	0.0187 NJ	0.00049 U	0.00049 U	0.0199 NJ
	3		0.000481 UJ	0.000481 UJ	0.000481 UJ	0.000481 UJ	0.000481 UJ
W002[b]	1	peristaltic	0.000481 UJ	0.000481 UJ	0.000481 UJ	0.000481 UJ	0.000481 UJ
	2		0.000495 U	0.000495 U	0.000495 U	0.000495 U	0.000495 U
	2 (FR)		NA	NA	NA	NA	NA
	3		0.000538 UJ	0.000538 UJ	0.000538 UJ	0.000538 UJ	0.000538 UJ
	3 (FR)		NA	NA	NA	NA	NA
W003	1	peristaltic	0.00049 UJ	0.00049 UJ	0.00049 UJ	0.00049 UJ	0.00049 UJ
	2		0.0005 U	0.0005 U	0.0005 U	0.0005 U	0.0005 U
	3		0.000505 UJ	0.000505 UJ	0.000505 UJ	0.000505 UJ	0.000505 UJ
W004	1	peristaltic	0.0005 U	0.0005 U	0.0005 U	0.0005 UJ	0.00050 U
	2		0.000485 U	0.000485 U	0.000485 U	0.000485 U	0.000485 U
	2 (FR)		0.000485 U	0.000485 U	0.000485 U	0.000485 U	0.000485 U
	3		0.000526 UJ	0.000526 UJ	0.000526 UJ	0.000526 UJ	0.000526 UJ
W006	1	peristaltic	0.0005 U	0.0005 U	0.0005 U	0.0005 UJ	0.00050 U
	2		0.00051 U	0.00051 U	0.00051 U	0.00051 U	0.00051 U
	3		0.000521 U	0.000521 U	0.000521 U	0.000521 UJ	0.000521 U
W007	1	peristaltic	0.000495 U	0.000495 U	0.000495 U	0.000495 UJ	0.000495 U
	2		0.0005 U	0.0005 U	0.0005 U	0.0005 U	0.00050 U
	3		0.000524 U	0.000524 U	0.000524 U	0.000524 UJ	0.000524 U
W008	1	peristaltic	0.000472 U	0.000472 U	0.000472 U	0.000472 UJ	0.000472 U
	2		0.00049 U	0.00049 U	0.00049 U	0.00049 U	0.00049 U
	3		0.000515 U	0.000515 U	0.000515 U	0.000515 UJ	0.000515 U
W009	1	peristaltic	0.0005 U	0.0005 U	0.0005 U	0.0005 U	0.00050 U
	2		0.000485 U	0.000485 U	0.000485 U	0.000485 U	0.000485 U
	3		0.000532 U	0.000532 U	0.000532 U	0.000532 UJ	0.000532 U
W012	1	peristaltic	0.0005 U	0.0005 U	0.0005 U	0.0005 U	0.00050 U
	2		0.00051 U	0.00051 U	0.00051 U	0.00051 U	0.00051 U
	3		0.000481 U	0.000481 U	0.000839	0.000481 U	0.00204

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Table 3-22. Chemical Concentrations for Round 2 COPCs (Pesticides) in Near-Bottom Surface Water

Location	Event[a]	Sampling Method	2,4′-DDD (µg/L)	2,4′-DDT (µg/L)	4,4′-DDD (µg/L)	4,4′-DDT (µg/L)	Total DDTs (µg/L)
W013-1	1	peristaltic	0.000472 U	0.000472 U	0.000472 U	0.000472 U	0.000472 U
	1	XAD	0.00000734 J	0.000001957 J	0.0000217	0.00000934 J	0.0000654 J
	2	peristaltic	NA	NA	NA	NA	NA
	2	XAD	0.00001178	0.00000349	0.0000351	0.00001244	0.0000931 J
	3	peristaltic	NA	NA	NA	NA	NA
	3	XAD	0.0000468	0.00000541 J	0.0001072	0.00001779	0.000241 J
W013-2[c]	1	peristaltic	0.000526 U	0.000526 U	0.000526 U	0.000526 U	0.000526 U
	1	XAD	0.00000566 J	0.000001672 J	0.00001713	0.00000831 J	0.0000552 J
	2	peristaltic	NA	NA	NA	NA	NA
	2	XAD	0.00000652	0.000000457 U	0.00001201	0.000001342 J	0.0000496 J
	3		0.000016347 J	0.0000011 J	0.00003812 J	0.00000279 J	0.0000857 J
W015	1	peristaltic	NA	NA	NA	NA	NA
	1	XAD	0.002057	0.000267	0.00311	0.001582	0.00766
	2	peristaltic	NA	NA	NA	NA	NA
	2	XAD	0.001134	0.000169	0.00238	0.001102	0.00521
	3	peristaltic	NA	NA	NA	NA	NA
	3	XAD	0.000883	0.0000333	0.00228	0.0000711	0.00359
W016-1	1	peristaltic	NA	NA	NA	NA	NA
	1	XAD	0.000391	0.0002134	0.000804	0.001338	0.00297
	2	peristaltic	NA	NA	NA	NA	NA
	2	XAD	0.0002379	0.0000824	0.00048	0.0002643	0.00124 J
	3	peristaltic	NA	NA	NA	NA	NA
	3	XAD	0.00137	0.000525	0.00325	0.003862	0.00976
W016-2	1	peristaltic	NA	NA	NA	NA	NA
W017	1	peristaltic	0.00051 U	0.00051 U	0.00051 U	0.00051 U	0.00051 U
	2		0.00051 U	0.00051 U	0.00051 U	0.00051 U	0.00051 U
	3		0.000481 U	0.000481 U	0.000481 U	0.000693 J	0.00190 J
W018[d]	1	peristaltic	0.000481 U	0.000481 U	0.000481 U	0.000481 U	0.000481 U
	1	XAD	0.0000068 U	0.00000103 J	0.0000209	0.000004054 J	0.0000610 J
	2	peristaltic	NA	NA	NA	NA	NA

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Table 3-22. Chemical Concentrations for Round 2 COPCs (Pesticides) in Near-Bottom Surface Water

Location	Event[a]	Sampling Method	2,4′-DDD (µg/L)	2,4′-DDT (µg/L)	4,4′-DDD (µg/L)	4,4′-DDT (µg/L)	Total DDTs (µg/L)
	2	XAD	0.00001381 J	0.000007887 J	0.0000358	0.00001278	0.000103 J
	3	peristaltic	NA	NA	NA	NA	NA
	3	XAD	0.00001377 J	0.000001195 J	0.00004365	0.00000511 J	0.0000999 J
W019	1	peristaltic	0.000481 U	0.000481 U	0.000481 U	0.000481 U	0.000481 U
	2		0.0005 U	0.0005 U	0.0005 U	0.00115	0.0024
	3		0.000485 U	0.000485 U	0.000485 U	0.000485 UJ	0.000485 U
W021	1	peristaltic	0.000481 U	0.000481 U	0.000481 U	0.000481 U	0.000481 U
	2		0.000481 U	0.000481 U	0.000481 U	0.000481 U	0.000481 U
	3		0.00049 U	0.00049 U	0.00049 U	0.00049 UJ	0.00049 U
W022	1	peristaltic	0.00049 U	0.00049 U	0.00049 U	0.00049 U	0.00049 U
	2		0.00049 U	0.00049 U	0.00049 U	0.00049 U	0.00049 U
	3		0.0005 U	0.0005 U	0.0005 U	0.0005 UJ	0.00050 U

[a] Sampling Event 1 – November 2004; Sampling Event 2 – March 2005; Sampling Event 3 – July 2005.
[b] PCB Aroclors were not analyzed at W002 during the November 2004 sampling event.
[c] There is no peristaltic data (only XAD data) for W013-2, Event 3.
[d] PCB Aroclors and pesticides were not analyzed at W018 during the March and July 2005 sampling events using the peristaltic method.
FR – field replicate
J – estimated concentration
N – presumptive evidence of a compound
NA – not analyzed
T – value calculated or selected from multiple results
U – not detected at given concentration

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Table 3-23. Selected Chronic and Acute Eco SLs for Benthic Invertebrate Round 2 COPCs

Round 2 COPC	Unit	Chronic Eco SL	Acute Eco SL	Source	Comments
Zinc (dissolved)	mg/L	0.0365	0.0362	AWQC	Criteria are hardness-dependent. These values correspond to a hardness of 25 mg/L calcium carbonate (as estimated for LWR) and were adjusted using EPA-provided equations. Criteria are for dissolved fraction.
Benzo(a)anthracene	µg/L	0.49	0.027	Tier II	
Benzo(a)pyrene	µg/L	0.24	0.014	Tier II	
4-Chloro-3-methylphenol	µg/L	0.6	30	ODEQ	Acute Eco SL was divided by a UF of 50 to calculate the chronic Eco SL.
Total PCBs	µg/L	0.014	2	ODEQ (acute); ODEQ and Tier II (chronic)	Total PCBs criterion applies to total PCBs, as either sum of all homologs, Aroclors, or congeners.
2,4′-DDD	µg/L	0.001	1.1	4,4′-DDT	AWQC for 4,4′-DDT applies to 2,4′-DDD.
2,4′-DDT	µg/L	0.001	1.1	4,4′-DDT	AWQC for 4,4′-DDT applies to 2,4′-DDT.
4,4′-DDD	µg/L	0.001	1.1	4,4′-DDT	AWQC for 4,4′-DDT applies to total 4,4′-DDD.
4,4′-DDT	µg/L	0.001	1.1	AWQC (acute), Tier II (chronic)	
Total DDTs	µg/L	0.001	1.1	AWQC (acute), Tier II (chronic)	AWQC for 4,4′-DDT applies to total DDTs.

AWQC – ambient water quality criteria
BCF – bioconcentration factor
COPC – chemical of potential concern
Eco SL – ecological screening level
EPA – US Environmental Protection Agency
LWR – Lower Willamette River
ODEQ – Oregon Department of Environmental Quality
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl

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Table 3-24. Round 2 COPCs Identified in TZW

Round 2 COPCs	
Metals	
Barium (total)[a]	Nickel (dissolved)[b]
Cadmium (dissolved)[b]	Sodium (total)[a]
Copper (dissolved)[b]	Vanadium (total)[a]
Lead (dissolved)[b]	Zinc (dissolved)[b]
PAHs	
2-Methylnaphthalene	Chrysene
Acenaphthene	Dibenzo(a,h)anthracene
Anthracene	Fluoranthene
Benzo(a)anthracene	Fluorene
Benzo(a)pyrene	Indeno(1,2,3-cd)pyrene
Benzo(b)fluoranthene	Naphthalene
Benzo(g,h,i)perylene	Phenanthrene
Benzo(k)fluoranthene	Pyrene
SVOCs	
1,2-Dichlorobenzene	Dibenzofuran
1,4-Dichlorobenzene	
Pesticides	
2,4′-DDD	4,4′-DDE
2,4′-DDT	4,4′-DDT
4,4′-DDD	Total DDTs
Herbicides	
Dalapon	Silvex
VOC	
1,1-Dichloroethene	Ethylbenzene
1,2,4-Trimethylbenzene	Isopropylbenzene
1,3,5-Trimethylbenzene	m,p-Xylene
Benzene	o-Xylene
Carbon disulfide	Toluene
Chlorobenzene	Total xylenes
Chloroethane	Trichloroethene
cis-1,2-Dichloroethene	Vinyl chloride
Cyanide	
Cyanide	
Perchlorate	
Perchlorate	

[a] Eco SL is based on a total concentration; dissolved concentrations were not evaluated.
[b] Eco SL is based on a dissolved concentration; total concentrations were not evaluated.

COPC – chemical of potential concern
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound
VOC – volatile organic compound

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Table 3-25. Results of TZW Framework for Identification of Potential TZW iCOCs

Round 2 COPC	Potential TZW iCOC?	Rationale for the Identificationof Potential iCOCs
Metals		
Barium (total)	no	Uniform concentration in sediment across the site.
Cadmium (dissolved)	no	Low exceedance ratios and frequencies.
Copper (dissolved)	no	Low exceedance ratios and frequencies.
Lead (dissolved)	no	Low exceedance ratios and frequencies.
Nickel (dissolved)	no	Low exceedance ratios and frequencies.
Sodium (total)	no	Limited spatial extent reduces relevance as a population- or community-level stressor. Associated with TZW at only two sites (Arkema chlorate plant and acid plant [mean HQ[a] = 2.0 and 21, respectively]), where it exceeded both chronic and acute Eco SLs. Not a potential iCOC based on other LOEs.
Vanadium (total)	no	Weak evidence; associated with TZW at one site (Siltronic [mean HQ[a] = 3.3]).
Zinc (dissolved)	no	Low exceedance ratios and frequencies.
PAHs		
2-Methylnaphthalene	yes	Relatively high exceedances of acute and chronic SLs at two sites (Gasco and Siltronic).
Acenaphthene	yes	Relatively high exceedances of acute and chronic SLs at two sites (Gasco and Siltronic).
Anthracene	yes	Relatively high exceedances of acute and chronic SLs at two sites (Gasco and Siltronic).
Benzo(a)anthracene	yes	Corroborated by total PAHs in multiple LOEs (laboratory-exposed worm tissue).
Benzo(a)pyrene	yes	Corroborated by total PAHs in multiple LOEs (laboratory-exposed worm tissue).
Benzo(b)fluoranthene	yes	Corroborated by total PAHs in multiple LOEs (laboratory-exposed worm tissue).
Benzo(g,h,i)perylene	yes	Corroborated by total PAHs in multiple LOEs (field-collected and laboratory-exposed clam tissue and laboratory-exposed worm tissue).
Benzo(k)fluoranthene	yes	Corroborated by total PAHs in multiple LOEs (laboratory-exposed worm tissue).
Chrysene	yes	Corroborated by total PAHs in multiple LOEs (laboratory-exposed worm tissue).
Dibenzo(a,h)anthracene	yes	Relatively high exceedances of acute and chronic SLs at two sites (Gasco and Siltronic).
Fluoranthene	yes	Relatively high exceedances of acute and chronic SLs at two sites (Gasco and Siltronic).
Fluorene	yes	Relatively high exceedances of acute and chronic SLs at three sites (ExxonMobil Oil, Gasco, and Siltronic).
Indeno(1,2,3-cd)pyrene	yes	Relatively high exceedances of acute and chronic SLs at three sites (ExxonMobil Oil, Gasco, and Siltronic).
Naphthalene	yes	Relatively high exceedances of acute and chronic SLs at three sites (Arkema acid plant, Gasco, and Siltronic).
Phenanthrene	yes	Relatively high exceedances of acute and chronic SLs at three sites (ExxonMobil Oil, Gasco, and Siltronic).
Pyrene	yes	Corroborated by total PAHs in multiple LOEs (laboratory-exposed worm tissue).

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Table 3-25. Results of TZW Framework for Identification of Potential TZW iCOCs

Round 2 COPC	Potential TZW iCOC?	Rationale for the Identificationof Potential iCOCs
SVOCs		
1,2-Dichlorobenzene	no	Associated with TZW at only one site (Rhône-Poulenc [mean HQ[a] = 7.0]), where it exceeded both chronic and acute Eco SLs. Not a potential iCOC based on other LOEs.
1,4-Dichlorobenzene	no	Associated with TZW at only one site (Rhône-Poulenc [mean HQ[a] = 2.3]), where it exceeded both chronic and acute Eco SLs. Not a potential iCOC based on other LOEs.
Dibenzofuran	no	Low exceedance ratios and frequencies.
Pesticides		
2,4′-DDD	yes	Corroborated by total DDTs in multiple LOEs (surface water).
2,4′-DDT	yes	Corroborated by total DDTs in multiple LOEs (surface water).
4,4′-DDD	yes	Corroborated by total DDTs in multiple LOEs (field-collected and laboratory-exposed clam tissue and laboratory-exposed worm tissue).
4,4′-DDE	yes	Corroborated by total DDTs in multiple LOEs.
4,4′-DDT	yes	Corroborated by total DDTs in multiple LOEs (surface water and field-collected clam tissue).
Total DDTs	yes	Corroborated by predictive model, field-collected and laboratory-exposed clam tissues, laboratory-exposed worm tissue, and surface water.
Herbicides		
Dalapon	no	Low exceedance ratios and frequencies (mean HQ[a] = 1.2).
Silvex	no	Low exceedance ratios and frequencies (mean HQ[a] = 0.32).
VOCs		
1,1-Dichloroethene	no	Low exceedance ratios.
1,2,4-Trimethylbenzene	no	Low exceedance ratios.
1,3,5-Trimethylbenzene	no	Low exceedance ratios.
Benzene	no	Exceeded both acute and chronic Eco SLs at one sampling station (Siltronic site [mean HQ[a] = 1.1]). Exceeded chronic Eco SL only at one sampling station (Gasco site [mean HQ[a] = 1.1]).
Carbon disulfide	no	Exceeded both acute and chronic Eco SLs at two sampling stations at one site (Gasco [mean HQ[a] = 120]). Exceeded chronic Eco SL only at one site (Siltronic [0.88]).
Chlorobenzene	no	Exceeded both acute and chronic Eco SLs at one sampling station (Arkema acid plant [mean HQ[a] = 27]). Exceeded chronic Eco SL only at one sampling station (Rhône-Poulenc [mean HQ[a] = 0.79]).
Chloroethane	no	Low exceedance ratios.
cis-1,2-Dichloroethene	no	Exceeded acute and chronic Eco SLs at two sampling stations at one site (Siltronic [mean HQ[a] = 2.8]).
Ethylbenzene	no	Exceeded chronic Eco SL by >10x at six sampling stations at Siltronic site (mean HQ[a] = 4.2) and one sampling station at Gasco site (mean HQ[a] = 2.6). No exceedances of acute Eco SL.

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Table 3-25. Results of TZW Framework for Identification of Potential TZW iCOCs

Round 2 COPC	Potential TZW iCOC?	Rationale for the Identificationof Potential iCOCs
Isopropylbenzene	no	Low exceedance ratios.
m,p-Xylene	no	Low exceedance ratios.
o-Xylene	no	Exceeded chronic Eco SL by > 10x at one sampling station at Siltronic site (mean HQ[a] = 0.76). Also exceeded chronic Eco SL at Gasco site (mean HQ[a] = 0.87). No exceedances of acute Eco SL.
Toluene	no	Exceeded chronic Eco SL by > 10x at one sampling station at Siltronic site (mean HQ[a] = 0.88). Also exceeded chronic Eco SL at Gasco site (mean HQ[a] = 0.93). No exceedances of acute Eco SL.
Total xylenes	no	Exceeded chronic Eco SL by > 10x at three sampling stations at one site (Siltronic [mean HQ[a] = 1.8]). Also exceeded chronic Eco SL at Gasco site (mean HQ[a] = 1.8). Exceeded acute Eco SL at one of the three sampling stations where it exceeded the chronic Eco SL by > 10x.
Trichloroethene	no	Low exceedance ratios.
Vinyl chloride	no	Low exceedance ratios.
Cyanide		
Cyanide	yes	Relatively high exceedances of chronic and acute Eco SLs at two sites (Gasco and Siltronic [mean HQs[a] = 1,070 and 32, respectively]).
Perchlorate		
Perchlorate	yes	Exceeded chronic Eco SL at two sites (Arkema acid plant [mean HQ[a] = 140] and Arkema chlordate plant [mean HQ[a] = 2,900]). Did not exceed either acute or chronic Eco SLs at Gunderson.

[a] All mean HQs are based on chronic criteria.

Eco SL – ecological screening level
iCOC – initial chemical of concern
ODEQ – Oregon Department of Environmental Quality
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound
TZW – transition zone water
VOC – volatile organic compound

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Table 3-26. Comparison of Metals Round 2 COPC Concentrations in Sediment to Upstream Background Concentrations in Sediment

Round 2 COPC	No. of Detections	No. of Detected Exceedances of Upstream Background Concentrations in Sediment	Percent Exceedance of Threshold Level in Detections
Cadmium	1,053	382	36%
Copper	1,139	526	46%
Lead	1,136	700	62%
Zinc	1,139	440	39%

UCL – upper confidence limit

Table 3-27. Potential TZW iCOCs

Potential TZW iCOCs	
PAHs	
2-Methylnaphthalene	Chrysene
Acenaphthene	Dibenzo(a,h)anthracene
Anthracene	Fluoranthene
Benzo(a)anthracene	Fluorene
Benzo(a)pyrene	Indeno(1,2,3-cd)pyrene
Benzo(b)fluoranthene	Naphthalene
Benzo(g,h,i)perylene	Phenanthrene
Benzo(k)fluoranthene	Pyrene
Pesticides	
2,4′-DDD	4,4′-DDE
2,4′-DDT	4,4′-DDT
4,4′-DDD	Total DDTs
Cyanide	
Cyanide	
Perchlorate	
Perchlorate	

iCOC – initial chemical of concern
TZW – transition zone water

Table 3-28. Eco SLs for the Potential TZW iCOCs

Potential iCOC	Chronic Eco SL (µg/L)	Acute Eco SL (µg/L)	Source	Comments
PAHs				
2-methylnaphthalene	2.1	37	Tier II	The Eco SLs for 1-methynapthalene were used as a surrogate.
Acenaphthene	23	80	Tier II	The acute and chronic Eco SLs are the EPA-calculated final acute value (FAV) and final chronic value (FCV), respectively, for sediment quality guideline development.
Anthracene	0.73	13	Tier II	Eco SLs were calculated for AWQC but never adopted.
Benzo(a)anthracene	0.027	0.49	Tier II	
Benzo(a)pyrene	0.014	0.24	Tier II	
Benzo(b)fluoranthene	0.6774	NA	EPA (2003)	Per EPA (2006), the acute Eco SL was not adopted.
Benzo(g,h,i)perylene	0.4391	NA	EPA (2003)	Per EPA (2006), the acute Eco SL was not adopted.
Benzo(k)fluoranthene	0.6415	NA	EPA (2003)	Per EPA (2006), the acute Eco SL was not adopted.
Chrysene	2.042	NA	EPA (2003)	Per EPA (2006), the acute Eco SL was not adopted.
Dibenzo(a,h)anthracene	0.2825	0.24	Tier II (acute); EPA (2003b) (chronic)	Acute Eco SL for benzo(a)pyrene was used as a surrogate.
Fluoranthene	6.16	33.6	Tier II	Eco SLs were calculated for AWQC, though were never adopted.
Fluorene	3.9	70	Tier II	
Indeno(1,2,3-cd)pyrene	0.275	0.24	Tier II (acute); EPA (2003b) (chronic)	Acute Eco SL for benzo(a)pyrene was used as a surrogate.
Naphthalene	12	190	Tier II	
Phenanthrene	6.3	30	Tier II	Eco SLs were calculated for AWQC but never adopted.
Pyrene	10.11	NA	EPA (2003)	Per EPA (2006), the acute Eco SL was not adopted.
Pesticides				
2,4′-DDD	0.001	1.1	AWQC	AWQC for 4,4′-DDT applies to 2,4′-DDD.
2,4′-DDT	0.001	1.1	AWQC	AWQC for 4,4′-DDT applies to 2,4′-DDT.
4,4′-DDD	0.001	1.1	AWQC	AWQC for 4,4′-DDT applies to 4,4′-DDD.
4,4′-DDE	0.001	1.1	AWQC	AWQC for 4,4′-DDT applies to 4,4′-DDE.
4,4′-DDT	0.001	1.1	AWQC	
Total DDTs	0.001	1.1	AWQC	AWQC for 4,4′-DDT applies to total DDTs.
Cyanide				
Cyanide	5.2	22	AWQC	
Perchlorate				
Perchlorate	NA	18	Goleman et al. (2002)	Chronic Eco SL was based on amphibian growth LOAEL following 70 days of exposure.

AWQC – ambient water quality criteria
Eco SL – ecological screening level
EPA – US Environmental Protection Agency
iCOC – initial chemical of concern
NA – not available
LWR – Lower Willamette River
ODEQ – Oregon Department of Environmental Quality

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Table 3-29. Comparison of EqP-Predicted TZW Concentrations to Eco SLs

Round 2 COPC	95th Percentile Sediment Concentration (mg/kg OC)	K_{oc}	Predicted TZW Concentration (µg/L)[a]	Chronic Eco SL (µg/L)	Acute Eco SL (µg/L)
Acenaphthylene	29.7	4,074	7.29	306.9	NA
Acetone	1.17	5.37	220	1,500	28,000

[a] TZW concentrations were predicted in the EqP approach by dividing the 95th OC-normalized sediment concentrations by the K_{OC} values.

COPC – chemical of potential concern
Eco SL – ecological screening level
EqP – equilibrium partitioning
NA – not available
TZW – transition zone water
OC – organic carbon

Table 3-30. Effects Level Designation of Sediment Samples Based on the Two Toxicity Tests

Toxicity Test Endpoint	Effect Level			
	No Effect	Not Reportable[a]	Minor Effect (Level 2)	Moderate Effect (Level 3)
Chironomus, mortality	174	0	34	25
Chironomus, growth	186	6	24	17
Hyalella, mortality	195	0	20	18
Hyalella, growth	85	4	98	46

[a] At four locations, all organisms (both *Chironomus* and *Hyalella*) died; and at two locations, all *Chironomus* died. Hence, no growth effects were reported at these locations.

Table 3-31. Overall Effects Level of the 233 Toxicity Test Locations

Overall Effects Level	Number of Locations
No effect (not toxic)	191
Level 2	10
Level 3	32

Table 3-32. FPM and LRM Sediment Quality Values

Round 2 COPC	Unit	FPM Level 2	FPM Level 3	LRM T35[a]	LRM T60[a]
Metals					
Antimony	mg/kg ww	NA	NA	1.6	3.1
Arsenic	mg/kg ww	16	16	8.6	26
Cadmium	mg/kg ww	3.6	3.6	0.35	1.4
Copper	mg/kg ww	NA	NA	230	480
Lead	mg/kg ww	1,300	1,300	52	330
Mercury	mg/kg ww	0.42	0.64	0.25	0.6
Silver	mg/kg ww	1.8	1.8	NA	NA
Zinc	mg/kg ww	700	700	240	530
Butyltins					
Tributyltin ion	µg/kg ww	NA	NA	60	510
PAHs					
Total PAHs	µg/kg ww	NA	NA	14,000	35,000
Phthalates					
Dibutyl phthalate	µg/kg ww	90	90	NA	NA
Phenols					
Phenol	µg/kg ww	NA	NA	60	160
PCBs					
Total PCBs	µg/kg ww	1,500	1,500	870	2,100
Pesticides					
Sum DDD	µg/kg ww	100	100	NA	NA
Sum DDE	µg/kg ww	1,020	1,020	NA	NA
Sum DDT	µg/kg ww	8,800	8,800	NA	NA
Total DDTs	µg/kg ww	NA	NA	390	1,000
Dieldrin	µg/kg ww	25	25	3.3	32
alpha-Chlordane	µg/kg ww	NA	NA	4	18
gamma-Chlordane	µg/kg ww	NA	NA	2	4.9
Total chlordanes	µg/kg ww	670	670	NA	NA
alpha-HCH	µg/kg ww	5.2	5.2	NA	NA
beta-HCH	µg/kg ww	11	11	3.7	15
delta-HCH	µg/kg ww	2.4	2.4	NA	NA
Endrin ketone	µg/kg ww	11	11	NA	NA
TPH					
Diesel-range hydrocarbons	mg/kg ww	345	345	120	210
Residual-range hydrocarbons	mg/kg ww	2,840	4,400	600	1,700
Conventionals					
Ammonia	mg/kg ww	170	280	80	130

Table 3-32. FPM and LRM Sediment Quality Values

Round 2 COPC	Unit	FPM Level 2	FPM Level 3	LRM T35[a]	LRM T60[a]
Sulfide	mg/kg ww	210	420	33	76
Percent fines	%	100	100	NA	NA

[a] Probabilities of toxicity of 0.35 and 0.60 through inversion of the logistic calibration equation for the multi-chemical model.

COPC – chemical of potential concern
FPM – floating percentile model
HCH – hexachlorocyclohexane
LRM – logistic regression model
NA – not available or not applicable
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TBT – tributyltin
TPH – total petroleum hydrocarbons
ww – wet weight

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Table 3-33. Decision Matrix for Toxicity Classification Predictions Based on the Two Models

		LRM		
		Not Toxic	**Indeterminate**	**Toxic**
FPM	**Not Toxic**	not toxic	not toxic	indeterminate
	Indeterminate	not toxic	indeterminate	toxic
	Toxic	indeterminate	toxic	toxic

FPM – floating percentile model
LRM – logistic regression model

Table 3-34. Decision Matrix with the ERA Surface Sediment Samples as Classified by the Two Models

		LRM[a]			
		Not Evaluated	**Not Toxic**	**Indeterminate**	**Toxic**
FPM[b]	**Not Evaluated**	16	32 (not toxic)	4 (indeterminate)	3 (toxic)
	Not Toxic	0	419 (not toxic)	253 (not toxic)	149 (indeterminate)
	Indeterminate	0	0 (not toxic)	4 (indeterminate)	40 (toxic)
	Toxic	0	10 (indeterminate)	20 (toxic)	97 (toxic)

[a] Predictions are based on a PrMax, calibrated to the Portland Harbor pooled Level 2 effects (all *Chironomus* and *Hyalella* endpoints). PrMax $\leq$ 0.40 is "not toxic" and PrMax > 0.61 is "toxic"; others are "indeterminate."

[b] Predictions are based on all endpoints/all chemicals below Level 2 thresholds classified as "not toxic"; any endpoint/any chemical above Level 3 thresholds classified as toxic"; otherwise "indeterminate."

FPM – floating percentile model
LRM – logistic regression model

Table 3-35. Potential iCOCs Based on FPM

Potential iCOCs
Metals
Arsenic
Cadmium
Lead
Mercury
Silver
Zinc
Phthalates
Dibutyl phthalate
PCBs
Total PCBs
Pesticides
Sum DDD
Sum DDE
Sum DDT
Dieldrin
Total chlordanes
alpha-HCH
beta-HCH
delta-HCH
Endrin ketone
TPH[a]
Diesel-range hydrocarbons
Residual-range hydrocarbons
Conventionals
Ammonia
Sulfide
Percent fines

[a] The use of TPH in the FPM is uncertain and will be re-evaluated in the BERA.

iCOC – initial chemical of concern
FPM – floating percentile model
HCH – hexachlorocyclohexane
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TPH – total petroleum hydrocarbons

Table 3-36. HQs for Round 2 COPCs Based on Field-Collected Clams

COPC	Range of HQs	Mean HQs	HQ Based on Site-Wide UCL
Metals[a]			
Cadmium	0.56 – **2.4**	1.0	**1.2**
Copper	**2.2 – 4.4**	**3.1**	**3.2**
Zinc	0.79 – **2.0**	**1.3**	**1.4**
Butyltins			
TBT ion	0.036 – **11**	0.58	**4.4**
PAHs			
Total PAHs	0.035 – **5.0**	0.62	0.89
Phthalates			
Dibutyl phthalate	0.059 – **4.8**	0.39	**1.9**
PCBs			
Total PCBs	0.070 – **3.7**	0.34	0.80
Pesticides			
4,4′-DDD	0.033 – **3.0**	0.38	**1.5**
Total DDTs	0.027 – **1.6**	0.20	0.45

[a] Five of the 33 tissue samples (BTFC011, BTFC018, BTFC029, BTFC032, and BTFC033) were not analyzed for metals because of limited biomass.

COPC – chemical of potential concern
HQ – hazard quotient
PAH – polycyclic aromatic hydrocarbons
PCB – polychlorinated biphenyls
TBT – tributyltin
UCL – upper confidence limit
Bold identifies HQs greater than 1.0.

Table 3-37. HQs for Round 2 COPCs Based on Field-Collected Crayfish and Invertebrates Collected with Multiplate Samplers

COPC	Range of HQs	Mean HQs	HQ Based on Site-Wide UCL
Crayfish			
Copper	**3.4 – 5.7**	**4.6**	**4.8**
Multiplate Invertebrates			
Copper	0.97 and **1.9**	**1.4**	NC

COPC – chemical of potential concern
HQ – hazard quotient
NC – not calculated
UCL – upper confidence limit
Bold identifies HQs greater than 1.0.

Table 3-38. HQs for Round 2 COPCs Based on Laboratory-Exposed Clams

Round 2 COPC	Range of HQs	Mean HQs	HQ Based on Site-Wide UCL
Copper	0.85 – **1.9**	**1.2**	**1.3**
TBT ion	0.018 – **14**	0.47	**4.2**
Total PAHs	0.019 – **1.3**	0.14	0.60
BEHP	0.14 – **22**	0.83	**3.6**
4,4′-DDD	0.0069 – **13**	0.44	**4.1**
Total DDTs	0.0039 – **3.6**	0.12	**1.1**

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
HQ – hazard quotient
PAH – polycyclic aromatic hydrocarbon
TBT – tributyltin
UCL – upper confidence limit
Bold identifies HQs greater than 1.0.

Table 3-39. HQs for Round 2 COPCs Identified Based on Laboratory-Exposed Worms

COPC	Range of HQs	Mean HQ	HQ Based on Site-Wide UCL
Metals			
Arsenic	0.17 – **1.8**	0.70	0.82
Cadmium	0.40 – **2.8**	0.77	0.92
Copper	0.59 – **6.5**	0.95	**1.2**
Zinc	0.67 – **1.2**	0.97	1.0
Butyltins			
TBT ion	0.0088 – **34**	1.1	**11**
PAHs			
Benzo(a)anthracene	0.0059 – **2.6**	0.20	1.0
Benzo(a)pyrene	0.0022 – **1.5**	0.10	0.61
Benzo(b)fluoranthene	0.0077 – **1.6**	0.15	0.76
Benzo(k)fluoranthene	0.0039 – **1.5**	0.11	0.61
Chrysene	0.014 – **3.9**	0.35	**1.7**
Pyrene	0.016 – **11**	0.87	**4.6**
Total PAHs	0.084 – **37**	3.1	**16**
Phthalates			
Dibutyl phthalate	0.059 – **1.7**	0.45	0.85
PCBs			
Total PCBs	0.062 – **6.0**	0.85	**3.3**
Pesticides			
4,4′-DDD	0.076 – **20**	1.3	**7.8**
Total DDTs	0.050 – **5.1**	0.41	**2.1**

COPC – chemical of potential concern
HQ – hazard quotient
PAH – polycyclic aromatic hydrocarbons
PCB – polychlorinated biphenyls
TBT – tributyltin
UCL – upper confidence limit
Bold identifies HQs greater than 1.0.

Table 3-40. UCL HQs for Round 2 COPCs Based on the Predicted Tissue Line of Evidence

COPC	UCL HQs		
	Field-Collected Clam	**Laboratory-Exposed Clam**	**Laboratory-Exposed Worm**
Antimony	NA	NA	0.84
Arsenic	NA	NA	0.79
Cadmium	**12**	NA	NA
Copper	NA	**1.3**	NA
Zinc	**1.3**	NA	NA
TBT ion	0.10[a]	0.15[b]	0.29[c]
Benzo(a)pyrene	NA	NA	**4.2**
Pyrene	NA	**1.4**	**3.8**
Total PAHs	**2.8**	**4.1**	**9.8**
beta-HCH	NA	**1.4**	0.53
Endrin	NA	**2.3**	**1.9**

[a] UCL HQ of 1.1 based on the aquatic TRV.
[b] UCL HQ of 1.8 based on the aquatic TRV.
[c] UCL HQ of 3.4 based on the aquatic TRV.
COPC – chemical of potential concern
HCH –hexachlorocyclohexane
HQ – hazard quotient
LOE – line of evidence
NA – not available
PAH – polycyclic aromatic hydrocarbon
TBT – tributyltin
UCL – upper confidence limit

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Table 3-41. Chronic and Acute HQs for Round 2 COPCs at 17 Locations During Three Surface Water Sampling Events

Round 2 COPCs by Sampling Location	Sampling Method	Event 1 (Nov 2004)		Event 2 (March 2005)		Event 3 (July 2005)	
		Chronic HQ	Acute HQ	Chronic HQ	Acute HQ	Chronic HQ	Acute HQ
W001							
Zinc (dissolved)	peristaltic	0.038	0.039	0.038	0.039	*0.033*	*0.033*
Benzo(a)anthracene	peristaltic	*0.31*	*0.017*	*0.16*	*0.0086*	*0.078*	*0.0043*
Benzo(a)pyrene	peristaltic	*0.46*	*0.027*	*0.23*	*0.013*	*0.11*	*0.0067*
4-Chloro-3-methylphenol	peristaltic	*0.048*	*0.00097*	*0.053*	*0.0011*	*0.08*	*0.0016*
Total PCBs	peristaltic	*0.18*	*0.0013*	*0.18*	*0.0013*	0.63	0.0044
2,4′-DDD	peristaltic	*0.47*	*0.00043*	*0.49*	*0.00045*	*0.48*	*0.00044*
2,4′-DDT	peristaltic	*0.47*	*0.00043*	**19**	0.017	*0.48*	*0.00044*
4,4′-DDD	peristaltic	*0.47*	*0.00043*	*0.49*	*0.00045*	*0.48*	*0.00044*
4,4′-DDT	peristaltic	*0.47*	*0.00043*	*0.49*	*0.00045*	*0.48*	*0.00044*
Total DDTs	peristaltic	*0.47*	*0.00043*	**20**	0.018	*0.48*	*0.00044*
W002							
Zinc (dissolved)	peristaltic	0.063	0.064	0.033	0.033	*0.032*	*0.032*
Benzo(a)anthracene	peristaltic	*0.078*	*0.0043*	*0.16*	*0.0086*	*0.078*	*0.0043*
Benzo(a)pyrene	peristaltic	*0.11*	*0.0067*	*0.23*	*0.013*	*0.11*	*0.0067*
4-Chloro-3-methylphenol	peristaltic	*0.048*	*0.00097*	*0.68*	*0.014*	*0.053*	*0.0011*
Total PCBs	peristaltic	NA	NA	*0.18*	*0.0013*	*0.18*	*0.0013*
2,4′-DDD	peristaltic	*0.48*	*0.00044*	*0.50*	*0.00045*	*0.54*	*0.00049*
2,4′-DDT	peristaltic	*0.48*	*0.00044*	*0.50*	*0.00045*	*0.54*	*0.00049*
4,4′-DDD	peristaltic	*0.48*	*0.00044*	*0.50*	*0.00045*	*0.54*	*0.00049*
4,4′-DDT	peristaltic	*0.48*	*0.00044*	*0.50*	*0.00045*	*0.54*	*0.00049*
Total DDTs	peristaltic	*0.48*	*0.00044*	*0.50*	*0.00045*	*0.54*	*0.00049*
W002 (FR)							
Zinc (dissolved)	peristaltic	NA	NA	0.030	0.030	*0.040*	*0.040*
Benzo(a)anthracene	peristaltic	NA	NA	*0.16*	*0.0086*	*0.078*	*0.0043*
Benzo(a)pyrene	peristaltic	NA	NA	*0.23*	*0.013*	*0.11*	*0.0067*
4-Chloro-3-methylphenol	peristaltic	NA	NA	*1.1*	*0.023*	*0.10*	*0.0021*

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Table 3-41. Chronic and Acute HQs for Round 2 COPCs at 17 Locations During Three Surface Water Sampling Events

Round 2 COPCs by Sampling Location	Sampling Method	Event 1 (Nov 2004)		Event 2 (March 2005)		Event 3 (July 2005)	
		Chronic HQ	Acute HQ	Chronic HQ	Acute HQ	Chronic HQ	Acute HQ
W003							
Zinc (dissolved)	peristaltic	0.025	0.025	0.049	0.050	0.052	0.052
Benzo(a)anthracene	peristaltic	*0.31*	*0.017*	*0.16*	*0.0086*	0.11	0.0061
Benzo(a)pyrene	peristaltic	*0.46*	*0.027*	*0.23*	*0.013*	0.19	0.011
4-Chloro-3-methylphenol	peristaltic	0.15	0.0030	**1.1**	0.022	*0.050*	*0.0010*
Total PCBs	peristaltic	*0.18*	*0.0013*	*0.18*	*0.0013*	*0.18*	*0.0013*
2,4′-DDD	peristaltic	*0.49*	*0.00045*	*0.50*	*0.00045*	*0.51*	*0.00046*
2,4′-DDT	peristaltic	*0.49*	*0.00045*	*0.50*	*0.00045*	*0.51*	*0.00046*
4,4′-DDD	peristaltic	*0.49*	*0.00045*	*0.50*	*0.00045*	*0.51*	*0.00046*
4,4′-DDT	peristaltic	*0.49*	*0.00045*	*0.50*	*0.00045*	*0.51*	*0.00046*
Total DDTs	peristaltic	*0.49*	*0.00045*	*0.50*	*0.00045*	*0.51*	*0.00046*
W004							
Zinc (dissolved)	peristaltic	0.068	0.069	0.12	0.12	0.052	0.052
Benzo(a)anthracene	peristaltic	0.44	0.024	*0.17*	*0.0096*	*0.078*	*0.0043*
Benzo(a)pyrene	peristaltic	*0.46*	*0.027*	*0.35*	*0.020*	*0.11*	*0.0067*
4-Chloro-3-methylphenol	peristaltic	*0.048*	*0.00097*	*0.050*	*0.0010*	*0.12*	*0.0025*
Total PCBs	peristaltic	0.60	0.0042	**1.2**	0.0081	*0.18*	*0.0013*
2,4′-DDD	peristaltic	*0.50*	*0.00045*	*0.49*	*0.00044*	*0.53*	*0.00048*
2,4′-DDT	peristaltic	*0.50*	*0.00045*	*0.49*	*0.00044*	*0.53*	*0.00048*
4,4′-DDD	peristaltic	*0.50*	*0.00045*	*0.49*	*0.00044*	*0.53*	*0.00048*
4,4′-DDT	peristaltic	*0.50*	*0.00045*	*0.49*	*0.00044*	*0.53*	*0.00048*
Total DDTs	peristaltic	*0.50*	*0.00045*	*0.49*	*0.00044*	*0.53*	*0.00048*
W004 (FR)							
Total PCBs	peristaltic	NA	NA	0.96	0.0068	NA	NA
2,4′-DDD	peristaltic	NA	NA	*0.49*	*0.00044*	NA	NA
2,4′-DDT	peristaltic	NA	NA	*0.49*	*0.00044*	NA	NA
4,4′-DDD	peristaltic	NA	NA	*0.49*	*0.00044*	NA	NA
4,4′-DDT	peristaltic	NA	NA	*0.49*	*0.00044*	NA	NA
Total DDTs	peristaltic	NA	NA	*0.49*	*0.00044*	NA	NA

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Table 3-41. Chronic and Acute HQs for Round 2 COPCs at 17 Locations During Three Surface Water Sampling Events

Round 2 COPCs by Sampling Location	Sampling Method	Event 1 (Nov 2004)		Event 2 (March 2005)		Event 3 (July 2005)	
		Chronic HQ	Acute HQ	Chronic HQ	Acute HQ	Chronic HQ	Acute HQ
W006							
Zinc (dissolved)	peristaltic	0.077	0.077	0.049	0.050	0.055	0.055
Benzo(a)anthracene	peristaltic	0.22	0.012	*0.16*	*0.0086*	*0.16*	*0.0086*
Benzo(a)pyrene	peristaltic	*0.23*	*0.013*	*0.23*	*0.013*	*0.23*	*0.013*
4-Chloro-3-methylphenol	peristaltic	0.11	0.0023	*0.88*	*0.018*	*0.052*	*0.0010*
Total PCBs	peristaltic	*0.18*	*0.0013*	*0.18*	*0.0013*	*0.18*	*0.0013*
2,4′-DDD	peristaltic	*0.50*	*0.00045*	*0.51*	*0.00046*	*0.52*	*0.00047*
2,4′-DDT	peristaltic	*0.50*	*0.00045*	*0.51*	*0.00046*	*0.52*	*0.00047*
4,4′-DDD	peristaltic	*0.50*	*0.00045*	*0.51*	*0.00046*	*0.52*	*0.00047*
4,4′-DDT	peristaltic	*0.50*	*0.00045*	*0.51*	*0.00046*	*0.52*	*0.00047*
Total DDTs	peristaltic	*0.50*	*0.00045*	*0.51*	*0.00046*	*0.52*	*0.00047*
W007							
Zinc (dissolved)	peristaltic	0.041	0.041	0.047	0.047	0.066	0.066
Benzo(a)anthracene	peristaltic	*0.16*	*0.0086*	*0.16*	*0.0086*	0.24	0.013
Benzo(a)pyrene	peristaltic	*0.23*	*0.013*	*0.23*	*0.013*	0.29	0.017
4-Chloro-3-methylphenol	peristaltic	0.14	0.0027	*0.052*	*0.0010*	*0.052*	*0.0010*
Total PCBs	peristaltic	*0.19*	*0.0013*	*0.19*	*0.0013*	*0.18*	*0.0013*
2,4′-DDD	peristaltic	*0.50*	*0.00045*	*0.50*	*0.00045*	*0.52*	*0.00048*
2,4′-DDT	peristaltic	*0.50*	*0.00045*	*0.50*	*0.00045*	*0.52*	*0.00048*
4,4′-DDD	peristaltic	*0.50*	*0.00045*	*0.50*	*0.00045*	*0.52*	*0.00048*
4,4′-DDT	peristaltic	*0.50*	*0.00045*	*0.50*	*0.00045*	*0.52*	*0.00048*
Total DDTs	peristaltic	*0.50*	*0.00045*	*0.50*	*0.00045*	*0.52*	*0.00048*
W008							
Zinc (dissolved)	peristaltic	0.052	0.052	0.12	0.12	0.13	0.13
Benzo(a)anthracene	peristaltic	0.17	0.0094	*0.16*	*0.0086*	0.29	0.016
Benzo(a)pyrene	peristaltic	0.24	0.014	*0.23*	*0.013*	0.54	0.031
4-Chloro-3-methylphenol	peristaltic	*0.22*	*0.0043*	*0.052*	*0.0010*	*0.053*	*0.0011*
Total PCBs	peristaltic	*0.18*	*0.0013*	*0.18*	*0.0013*	*0.18*	*0.0013*
2,4′-DDD	peristaltic	*0.47*	*0.00043*	*0.49*	*0.00045*	*0.52*	*0.00047*

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Table 3-41. Chronic and Acute HQs for Round 2 COPCs at 17 Locations During Three Surface Water Sampling Events

Round 2 COPCs by Sampling Location	Sampling Method	Event 1 (Nov 2004)		Event 2 (March 2005)		Event 3 (July 2005)	
		Chronic HQ	Acute HQ	Chronic HQ	Acute HQ	Chronic HQ	Acute HQ
2,4′-DDT	peristaltic	*0.47*	*0.00043*	*0.49*	*0.00045*	*0.52*	*0.00047*
4,4′-DDD	peristaltic	*0.47*	*0.00043*	*0.49*	*0.00045*	*0.52*	*0.00047*
4,4′-DDT	peristaltic	*0.47*	*0.00043*	*0.49*	*0.00045*	*0.52*	*0.00047*
Total DDTs	peristaltic	*0.47*	*0.00043*	*0.49*	*0.00045*	*0.52*	*0.00047*
W009							
Zinc (dissolved)	peristaltic	0.047	0.047	0.044	0.044	0.049	0.050
Benzo(a)anthracene	peristaltic	0.16	0.0088	*0.16*	*0.0086*	*0.16*	*0.0086*
Benzo(a)pyrene	peristaltic	*0.23*	*0.013*	*0.23*	*0.013*	*0.23*	*0.013*
4-Chloro-3-methylphenol	peristaltic	*0.12*	*0.0023*	*0.050*	*0.0010*	*0.05*	*0.0010*
Total PCBs	peristaltic	*0.18*	*0.0013*	*0.18*	*0.0013*	*0.18*	*0.0013*
2,4′-DDD	peristaltic	*0.50*	*0.00045*	*0.49*	*0.00044*	*0.53*	*0.00048*
2,4′-DDT	peristaltic	*0.50*	*0.00045*	*0.49*	*0.00044*	*0.53*	*0.00048*
4,4′-DDD	peristaltic	*0.50*	*0.00045*	*0.49*	*0.00044*	*0.53*	*0.00048*
4,4′-DDT	peristaltic	*0.50*	*0.00045*	*0.49*	*0.00044*	*0.53*	*0.00048*
Total DDTs	peristaltic	*0.50*	*0.00045*	*0.49*	*0.00044*	*0.53*	*0.00048*
W012							
Zinc (dissolved)	peristaltic	0.041	0.041	0.047	0.047	0.049	0.050
Benzo(a)anthracene	peristaltic	0.70	0.039	*0.22*	*0.012*	**4.1**	0.22
Benzo(a)pyrene	peristaltic	**1.4**	0.083	*0.59*	*0.034*	**11**	0.63
4-Chloro-3-methylphenol	peristaltic	*0.085*	*0.0017*	*0.052*	*0.0010*	*0.25*	*0.0050*
Total PCBs	peristaltic	*0.18*	*0.0013*	*0.19*	*0.0013*	*0.18*	*0.0013*
2,4′-DDD	peristaltic	*0.50*	*0.00045*	*0.51*	*0.00046*	*0.48*	*0.00044*
2,4′-DDT	peristaltic	*0.50*	*0.00045*	*0.51*	*0.00046*	*0.48*	*0.00044*
4,4′-DDD	peristaltic	*0.50*	*0.00045*	*0.51*	*0.00046*	0.84	0.00076
4,4′-DDT	peristaltic	*0.50*	*0.00045*	*0.51*	*0.00046*	*0.48*	*0.00044*
Total DDTs	peristaltic	*0.50*	*0.00045*	*0.51*	*0.00046*	**2.0**	0.0019
W013-1							
Zinc (dissolved)	peristaltic	0.025	0.025	0.038	0.039	*0.045*	*0.045*

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Table 3-41. Chronic and Acute HQs for Round 2 COPCs at 17 Locations During Three Surface Water Sampling Events

Round 2 COPCs by Sampling Location	Sampling Method	Event 1 (Nov 2004)		Event 2 (March 2005)		Event 3 (July 2005)	
		Chronic HQ	Acute HQ	Chronic HQ	Acute HQ	Chronic HQ	Acute HQ
Benzo(a)anthracene	peristaltic	*0.31*	*0.017*	*0.078*	*0.0043*	*0.078*	*0.0043*
	XAD	0.020	0.0011	0.021	0.0012	0.037	0.0020
Benzo(a)pyrene	peristaltic	*0.46*	*0.027*	*0.11*	*0.0067*	*0.11*	*0.0067*
	XAD	0.030	0.0018	0.0013	0.000076	0.060	0.0035
4-Chloro-3-methylphenol	peristaltic	*0.048*	*0.00097*	0.12	0.0025	*0.055*	*0.0011*
Total PCBs	XAD	0.24	0.0017	0.86	0.0060	0.57	0.0040
2,4′-DDD	peristaltic	*0.47*	*0.00043*	NA	NA	NA	NA
	XAD	0.0073	0.0000067	0.012	0.000011	0.047	0.000043
2,4′-DDT	peristaltic	*0.47*	*0.00043*	NA	NA	NA	NA
	XAD	0.0020	0.0000018	0.0035	0.0000032	0.0054	0.0000049
4,4′-DDD	peristaltic	*0.47*	*0.00043*	NA	NA	NA	NA
	XAD	0.022	0.000020	0.035	0.000032	0.11	0.000097
4,4′-DDT	peristaltic	*0.47*	*0.00043*	NA	NA	NA	NA
	XAD	0.0093	0.0000085	0.012	0.000011	0.018	0.000016
Total DDTs	peristaltic	*0.47*	*0.00043*	NA	NA	NA	NA
	XAD	0.065	0.000059	0.093	0.000085	0.24	0.00022
W013-2							
Zinc (dissolved)	peristaltic	NA	NA	0.049	0.05	NA	NA
Benzo(a)anthracene	peristaltic	NA	NA	0.10	0.0057	NA	NA
Benzo(a)pyrene	peristaltic	NA	NA	0.17	0.010	NA	NA
4-Chloro-3-methylphenol	peristaltic	NA	NA	NA	NA	NA	NA
W013-2 (FR)							
Zinc (dissolved)	peristaltic	*0.06*	*0.061*	NA	NA	NA	NA
Benzo(a)anthracene	peristaltic	*0.31*	*0.017*	NA	NA	NA	NA
	XAD	0.016	0.00088	0.0044	0.00024	0.020	0.0011
Benzo(a)pyrene	peristaltic	*0.46*	*0.027*	NA	NA	NA	NA
	XAD	0.024	0.0014	0.0023	0.00013	*0.017*	*0.00097*
4-Chloro-3-methylphenol	peristaltic	*0.048*	*0.00097*	NA	NA	NA	NA
Total PCBs	XAD	0.18	0.0013	0.15	0.0010	0.15	0.0011

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Table 3-41. Chronic and Acute HQs for Round 2 COPCs at 17 Locations During Three Surface Water Sampling Events

Round 2 COPCs by Sampling Location	Sampling Method	Event 1 (Nov 2004)		Event 2 (March 2005)		Event 3 (July 2005)	
		Chronic HQ	Acute HQ	Chronic HQ	Acute HQ	Chronic HQ	Acute HQ
2,4′-DDD	peristaltic	*0.53*	*0.00048*	NA	NA	NA	NA
	XAD	0.0057	0.0000051	0.0065	0.0000059	0.016	0.000015
2,4′-DDT	peristaltic	*0.53*	*0.00048*	NA	NA	NA	NA
	XAD	0.0017	0.0000015	*0.00046*	*0.00000042*	0.0011	0.0000010
4,4′-DDD	peristaltic	*0.53*	*0.00048*	NA	NA	NA	NA
	XAD	0.017	0.000016	0.012	0.000011	0.038	0.000035
4,4′-DDT	peristaltic	*0.53*	*0.00048*	NA	NA	NA	NA
	XAD	0.0083	0.0000076	0.0013	0.0000012	0.0028	0.0000025
Total DDTs	peristaltic	*0.53*	*0.00048*	NA	NA	NA	NA
	XAD	0.055	0.00005	0.050	0.000045	0.086	0.000078
W015							
Zinc (dissolved)	peristaltic	0.060	0.061	0.049	0.050	0.044	0.044
Benzo(a)anthracene	peristaltic	*0.16*	*0.0086*	*0.16*	*0.0086*	*0.16*	*0.0086*
	XAD	0.12	0.0068	0.059	0.0033	0.23	0.013
Benzo(a)pyrene	peristaltic	*0.23*	*0.013*	*0.23*	*0.013*	*0.23*	*0.013*
	XAD	0.22	0.013	0.13	0.0073	0.47	0.028
4-Chloro-3-methylphenol	peristaltic	*0.20*	*0.0040*	*0.052*	*0.0010*	*0.23*	*0.0047*
Total PCBs	XAD	0.14	0.00097	0.039	0.00027	0.058	0.00040
2,4′-DDD	XAD	**2.1**	0.0019	**1.1**	0.0010	0.88	0.00080
2,4′-DDT	XAD	0.27	0.00024	0.17	0.00015	0.033	0.000030
4,4′-DDD	XAD	**3.1**	0.0028	**2.4**	0.0022	**2.3**	0.0021
4,4′-DDT	XAD	**1.6**	0.0014	**1.1**	0.0010	0.071	0.000065
Total DDTs	XAD	**7.7**	0.0070	**5.2**	0.0047	**3.6**	0.0033
W016-1							
Zinc (dissolved)	peristaltic	*0.038*	*0.039*	0.047	0.047	0.059	0.059
Benzo(a)anthracene	peristaltic	*0.16*	*0.0086*	*0.16*	*0.0086*	*0.16*	*0.0086*
	XAD	0.032	0.0017	0.024	0.0013	0.14	0.0076
Benzo(a)pyrene	peristaltic	*0.23*	*0.013*	*0.23*	*0.013*	*0.23*	*0.013*
	XAD	0.063	0.0037	0.056	0.0033	0.18	0.011

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Table 3-41. Chronic and Acute HQs for Round 2 COPCs at 17 Locations During Three Surface Water Sampling Events

Round 2 COPCs by Sampling Location	Sampling Method	Event 1 (Nov 2004)		Event 2 (March 2005)		Event 3 (July 2005)	
		Chronic HQ	Acute HQ	Chronic HQ	Acute HQ	Chronic HQ	Acute HQ
4-Chloro-3-methylphenol	peristaltic	*0.15*	*0.0029*	*0.050*	*0.0010*	*0.11*	*0.0022*
Total PCBs	XAD	0.027	0.00019	0.032	0.00022	0.091	0.00064
2,4′-DDD	XAD	0.39	0.00036	0.24	0.00022	**1.4**	0.0012
2,4′-DDT	XAD	0.21	0.00019	0.082	0.000075	0.53	0.00048
4,4′-DDD	XAD	0.80	0.00073	0.48	0.00044	**3.3**	0.0030
4,4′-DDT	XAD	**1.3**	0.0012	0.26	0.00024	**3.9**	0.0035
Total DDTs	XAD	**3.0**	0.0027	**1.2**	0.0011	**9.8**	0.0089
W016-2 (FR)							
Zinc (dissolved)	peristaltic	*0.044*	*0.044*	NA	NA	NA	NA
Benzo(a)anthracene	peristaltic	*0.16*	*0.0086*	NA	NA	NA	NA
Benzo(a)pyrene	peristaltic	*0.23*	*0.013*	NA	NA	NA	NA
4-Chloro-3-methylphenol	peristaltic	*0.052*	*0.0010*	NA	NA	NA	NA
W017							
Zinc (dissolved)	peristaltic	0.044	0.044	0.047	0.047	0.049	0.050
Benzo(a)anthracene	peristaltic	*0.078*	*0.0043*	*0.16*	*0.0086*	*0.16*	*0.0086*
Benzo(a)pyrene	peristaltic	*0.11*	*0.0067*	*0.23*	*0.013*	*0.23*	*0.013*
4-Chloro-3-methylphenol	peristaltic	*0.073*	*0.0015*	*0.052*	*0.0010*	*0.11*	*0.0021*
Total PCBs	peristaltic	NA	NA	*0.18*	*0.0013*	*0.18*	*0.0013*
2,4′-DDD	peristaltic	*0.51*	*0.00046*	*0.51*	*0.00046*	*0.48*	*0.00044*
2,4′-DDT	peristaltic	*0.51*	*0.00046*	*0.51*	*0.00046*	*0.48*	*0.00044*
4,4′-DDD	peristaltic	*0.51*	*0.00046*	*0.51*	*0.00046*	*0.48*	*0.00044*
4,4′-DDT	peristaltic	*0.51*	*0.00046*	*0.51*	*0.00046*	0.69	0.00063
Total DDTs	peristaltic	*0.51*	*0.00046*	*0.51*	*0.00046*	**1.9**	0.0017
W018							
Zinc (dissolved)	peristaltic	0.13	0.14	0.063	0.064	0.073	0.074
Benzo(a)anthracene	peristaltic	*0.31*	*0.017*	*0.078*	*0.0043*	*0.16*	*0.0086*
	XAD	0.014	0.00078	0.058	0.0032	0.0062	0.00034
Benzo(a)pyrene	peristaltic	*0.46*	*0.027*	*0.11*	*0.0067*	*0.23*	*0.013*
	XAD	0.027	0.0016	0.12	0.0067	0.0033	0.00019

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Table 3-41. Chronic and Acute HQs for Round 2 COPCs at 17 Locations During Three Surface Water Sampling Events

Round 2 COPCs by Sampling Location	Sampling Method	Event 1 (Nov 2004)		Event 2 (March 2005)		Event 3 (July 2005)	
		Chronic HQ	Acute HQ	Chronic HQ	Acute HQ	Chronic HQ	Acute HQ
4-Chloro-3-methylphenol	peristaltic	*0.048*	*0.00097*	0.53	0.011	*0.052*	*0.0010*
Total PCBs	XAD	0.031	0.00022	0.12	0.00084	0.059	0.00042
2,4′-DDD	peristaltic	*0.48*	*0.00044*	NA	NA	NA	NA
	XAD	0.0068	0.0000062	0.014	0.000013	0.014	0.000013
2,4′-DDT	peristaltic	*0.48*	*0.00044*	NA	NA	NA	NA
	XAD	0.0010	0.00000094	0.0079	0.0000072	0.0012	0.0000011
4,4′-DDD	peristaltic	*0.48*	*0.00044*	NA	NA	NA	NA
	XAD	0.021	0.000019	0.036	0.000033	0.044	0.000040
4,4′-DDT	peristaltic	*0.48*	*0.00044*	NA	NA	NA	NA
	XAD	0.0041	0.0000037	0.013	0.000012	0.0051	0.0000046
Total DDTs	peristaltic	*0.48*	*0.00044*	NA	NA	NA	NA
	XAD	0.061	0.000055	0.10	0.000094	0.10	0.000091
W019							
Zinc (dissolved)	peristaltic	0.044	0.044	0.047	0.047	0.048	0.048
Benzo(a)anthracene	peristaltic	*0.16*	*0.0086*	*0.16*	*0.0086*	*0.16*	*0.0086*
Benzo(a)pyrene	peristaltic	*0.23*	*0.013*	*0.23*	*0.013*	*0.23*	*0.013*
4-Chloro-3-methylphenol	peristaltic	*0.068*	*0.0014*	*0.052*	*0.0010*	*0.052*	*0.0010*
Total PCBs	peristaltic	*0.18*	*0.0013*	*0.19*	*0.0013*	*0.18*	*0.0013*
2,4′-DDD	peristaltic	*0.48*	*0.00044*	*0.50*	*0.00045*	*0.49*	*0.00044*
2,4′-DDT	peristaltic	*0.48*	*0.00044*	*0.50*	*0.00045*	*0.49*	*0.00044*
4,4′-DDD	peristaltic	*0.48*	*0.00044*	*0.50*	*0.00045*	*0.49*	*0.00044*
4,4′-DDT	peristaltic	*0.48*	*0.00044*	**1.2**	0.0010	*0.49*	*0.00044*
Total DDTs	peristaltic	*0.48*	*0.00044*	**2.4**	0.0022	*0.49*	*0.00044*
W021							
Zinc (dissolved)	peristaltic	0.088	0.088	0.055	0.055	0.044	0.044
Benzo(a)anthracene	peristaltic	*0.16*	*0.0086*	*0.16*	*0.0086*	*0.16*	*0.0086*
Benzo(a)pyrene	peristaltic	*0.23*	*0.013*	*0.23*	*0.013*	*0.23*	*0.013*
4-Chloro-3-methylphenol	peristaltic	*0.057*	*0.0011*	*0.053*	*0.0011*	*0.052*	*0.0010*
Total PCBs	peristaltic	*0.18*	*0.0013*	*0.19*	*0.0013*	*0.18*	*0.0013*

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Table 3-41. Chronic and Acute HQs for Round 2 COPCs at 17 Locations During Three Surface Water Sampling Events

Round 2 COPCs by Sampling Location	Sampling Method	Event 1 (Nov 2004)		Event 2 (March 2005)		Event 3 (July 2005)	
		Chronic HQ	Acute HQ	Chronic HQ	Acute HQ	Chronic HQ	Acute HQ
2,4′-DDD	peristaltic	*0.48*	*0.00044*	*0.48*	*0.00044*	*0.49*	*0.00045*
2,4′-DDT	peristaltic	*0.48*	*0.00044*	*0.48*	*0.00044*	*0.49*	*0.00045*
4,4′-DDD	peristaltic	*0.48*	*0.00044*	*0.48*	*0.00044*	*0.49*	*0.00045*
4,4′-DDT	peristaltic	*0.48*	*0.00044*	*0.48*	*0.00044*	*0.49*	*0.00045*
Total DDTs	peristaltic	*0.48*	*0.00044*	*0.48*	*0.00044*	*0.49*	*0.00045*
W022							
Zinc (dissolved)	peristaltic	**1.1**[a]	**1.2**[a]	0.10	0.10	0.074	0.075
Benzo(a)anthracene	peristaltic	*0.16*	*0.0086*	*0.16*	*0.0086*	*0.16*	*0.0086*
Benzo(a)pyrene	peristaltic	*0.23*	*0.013*	*0.23*	*0.013*	*0.23*	*0.013*
4-Chloro-3-methylphenol	peristaltic	*0.077*	*0.0015*	*0.048*	*0.00097*	*0.050*	*0.0010*
Total PCBs	peristaltic	*0.18*	*0.0013*	*0.18*	*0.0013*	0.51	0.0036
2,4′-DDD	peristaltic	*0.49*	*0.00045*	*0.49*	*0.00045*	*0.50*	*0.00045*
2,4′-DDT	peristaltic	*0.49*	*0.00045*	*0.49*	*0.00045*	*0.50*	*0.00045*
4,4′-DDD	peristaltic	*0.49*	*0.00045*	*0.49*	*0.00045*	*0.50*	*0.00045*
4,4′-DDT	peristaltic	*0.49*	*0.00045*	*0.49*	*0.00045*	*0.50*	*0.00045*
Total DDTs	peristaltic	*0.49*	*0.00045*	*0.49*	*0.00045*	*0.50*	*0.00045*

[a] For zinc, the acute Eco SL was lower than the chronic Eco SL as recommended by EPA.
HQ – hazard quotient
NA – not analyzed
PCB – polychlorinated biphenyl
XAD – Infiltrex 300 system with XAD-2 resin column
Regular font identifies HQs calculated using detected concentrations.
Italics identifies HQs calculated using non-detected concentrations.
Bold identifies HQs greater than 1.0.

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Table 3-42. Potential iCOCs Identified in Near-Bottom Surface Water

Chemical	Sampling Method	Unit	Minimum HQ	Maximum HQ	UCL HQ
Zinc	peristaltic	mg/L	0.015	**1.2**	0.15
Benzo(a)anthracene	peristaltic	µg/L	0.039	**4.1**	0.48
	XAD	µg/L	0.0013	0.15	0.061
Benzo(a)pyrene	peristaltic	µg/L	0.057	**11**	**1.1**
	XAD	µg/L	0.000678	0.25	0.097
4-chloro-3-methyl-phenol	peristaltic	µg/L	0.024	**1.1**	0.20
Total PCBs	peristaltic	µg/L	0.089	**1.3**	0.36
	XAD	µg/L	0.0123	0.86	0.31
2,4′-DDD	peristaltic	µg/L	0.24	0.27	0.25
	XAD	µg/L	0.00326	**2.1**	**2.1**
2,4′-DDT	peristaltic	µg/L	0.24	**19**	**2.5**
	XAD	µg/L	0.000229	0.53	1.0
4,4′-DDD	peristaltic	µg/L	0.24	0.84	0.29
	XAD	µg/L	0.012	**3.3**	**4.0**
4,4′-DDT	peristaltic	µg/L	0.24	**1.2**	0.32
	XAD	µg/L	0.00134	**3.9**	**5.9**
Total DDTs	peristaltic	µg/L	0.24	**20**	**2.8**
	XAD	µg/L	0.0431	**9.8**	**10**

PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
UCL – upper confidence limit
HQ – hazard quotient
XAD – Infiltrex 300 system with XAD-2 resin column
Bold identifies HQs greater than 1.0.

Table 3-43. Summary of HQs for Potential iCOCs for TZW

Analyte	Number of Samples	Number of Detects	Exceedance of Chronic SL				Exceedance of Acute SL			
			Number of Detected Exceedances	Minimum HQ	Maximum HQ	Mean HQ	Number of Detected Exceedances	Minimum HQ	Maximum HQ	Mean HQ
PAHs										
2-Methylnaphthalene	93	27	11	0.0016	40	1.7	5	0.000092	2.3	0.10
Acenaphthene	106	100	24	0.00010	17	1.1	10	0.000030	5.0	0.32
Anthracene	106	76	28	0.0018	87	3.3	5	0.00010	4.9	0.19
Benzo(a)anthracene	106	44	31	0.089	1,200	36	14	0.0049	66	2.0
Benzo(a)pyrene	106	38	34	0.13	2,700	77	15	0.0075	160	4.5
Benzo(b)fluoranthene	106	31	13	0.0032	49	1.4	NA	NA	NA	NA
Benzo(g,h,i)perylene	106	40	13	0.0093	66	1.8	NA	NA	NA	NA
Benzo(k)fluoranthene	106	27	10	0.0023	14	0.48	NA	NA	NA	NA
Chrysene	106	46	10	0.00069	17	0.50	NA	NA	NA	NA
Dibenzo(a,h)anthracene	106	30	8	0.0064	13	0.38	NA	NA	NA	NA
Fluoranthene	106	67	11	0.0011	17	0.71	4	0.00021	3.2	0.13
Fluorene	106	85	36	0.00079	28	2.1	2	0.000044	1.5	0.12
Indeno(1,2,3-cd)pyrene	106	39	13	0.0084	61	2.0	NA	NA	NA	NA
Naphthalene	173	75	31	0.00053	1,100	45	18	0.000033	72	2.9
Phenanthrene	106	71	36	0.00063	57	2.7	13	0.00013	12	0.56
Pyrene	106	72	11	0.00098	15	0.59	NA	NA	NA	NA
Pesticides										
2,4′-DDD	14	10	10	3.3	1,100	160	0	0.0030	1.0	0.14
2,4′-DDT	14	3	3	0.89	150	27	0	0.00081	0.14	0.025
4,4′-DDD	14	6	6	4.7	1,300	210	1	0.0043	1.2	0.20
4,4′-DDE	14	3	3	3.9	930	91	0	0.0035	0.85	0.083
4,4′-DDT	14	3	3	5.0	1,800	290	1	0.0045	1.6	0.26
Total DDTs	14	10	10	7.7	3,100	700	4	0.0070	2.8	0.64
Cyanide										
Cyanide	34	32	32	1.2	4,400	280	25	0.27	1100	67
Perchlorate										
Perchlorate	21	11	11	0.22	9,800	280	NA	NA	NA	NA

HQ – hazard quotient
iCOC – initial chemical of concern
NA – not applicable
PAH – polycyclic aromatic hydrocarbon
SL – screening level
TZW – transition zone water

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Table 3-44. Comparison of Selected Aquatic TRV with Invertebrate Effects Data Available in the Literature

Potential iCOC	Aquatic TRV	Unit (ww)	Invertebrate Effects Data	Source
Cadmium	0.09	mg/kg	Aquatic TRV is lower than all NOAELs reported in 13 studies with a factor ranging from 5 and to 5,900 (approximately 6 orders of magnitude).	Windward (2005)
Copper	3.1	mg/kg	Aquatic TRV is lower than all NOAELs reported six studies with a factor ranging from 2 to 16. However, it should be noted that all invertebrate studies evaluated adverse effects on survival.	Dyer et al. (2000)
Zinc	27	mg/kg	Aquatic TRV is within the concentration range of the NOAEL and LOAEL reported in one study with crayfish.	Dyer et al. (2000)
TBT	49.9	µg/kg	Aquatic TRV is approximately four times lower than the lowest NOAEL reported in five studies with marine invertebrates.	estimated from AWQC and BCF
Chrysene	1,000	µg/kg	Aquatic TRVs of 1,000 µg/kg ww for the six individual PAHs and total PAHs were within the concentration range of LOAELs and NOAELs reported in three studies with individual PAHs or a mixture of PAHs.	estimated from AWQC and BCF
Pyrene	1,000	µg/kg	Aquatic TRVs of 1,000 µg/kg ww for the six individual PAHs and total PAHs were within the concentration range of LOAELs and NOAELs reported in three studies with individual PAHs or a mixture of PAHs.	estimated from AWQC and BCF
Total PAHs	1,000	µg/kg	Aquatic TRVs of 1,000 µg/kg ww for the six individual PAHs and total PAHs were within the concentration range of LOAELs and NOAELs reported in three studies with individual PAHs or a mixture of PAHs.	estimated from AWQC and BCF
BEHP	390	µg/kg	Aquatic TRV is in the lower range of NOAELs for crustaceans reported in one study and in a compilation of studies.	Windward (2005)
Dibutyl phthalate	270	µg/kg	Aquatic TRV is just below the range of NOAELs for crustaceans reported in one study and in a compilation of studies.	estimated from AWQC and BCF
Total PCBs	720	µg/kg	Aquatic TRV is lower than all NOAELs reported for in nine studies with a factor ranging from 1.3 to 32.	Windward (2005)
4,4′-DDD	54	µg/kg	Aquatic TRV is lower than concentrations associated with adverse effects to amphipods reported in one study with a factor ranging from 278 to 2,148 (two orders of magnitude).	estimated from AWQC and BCF
Total DDTs	290	µg/kg	No invertebrate studies with total DDTs were available in the literature. The reported LOAELs and NOAELs for the isomer 4,4′-DDT span a large range of concentrations (approximately two order of magnitude). The aquatic TRV for falls within this range of concentrations.	Windward (2005)

AWQC – ambient water quality criteria
BCF – bioconcentration factor
BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
Eco SL – ecological screening level
LOAEL – lowest observed adverse effects level
NOAEL – no observed adverse effects level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TBT – tributyltin
TRV – toxicity reference value

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Table 4-1. Fish Known to be Present or Common in the LWR

Species	Scientific Name	Resident or Migratory[a]
Herbivore		
Chiselmouth	*Acrocheilus alutaceus*	resident
Mountain sucker	*Catostomus platyrhynchus*	resident
Omnivore		
Eulachon	*Thaleichthys pacificus*	migratory
Bluegill	*Lepomis macrochirus*	resident
Common carp	*Cyprinus carpio*	resident
Pumpkinseed	*Lepomis gibbosus*	resident
Largescale sucker	*Catostomus macrocheilus*	resident
Brown bullhead	*Ameiurus nebulosus*	resident
Goldfish	*Carassius auratus*	resident
Green sturgeon[b]	*Acipenser medirostris*	migratory
White sturgeon	*Acipenser transmontanus*	migratory
Yellow bullhead	*Ameiurus natalis*	resident
Invertivore		
American shad	*Alosa sapidissima*	migratory
Chinook salmon[c]	*Oncorhynchus tshawytscha*	migratory
Coastal cutthroat trout[d]	*Oncorhynchus clarki clarki*	migratory
Coho salmon[c]	*Oncorhynchus kisutch*	migratory
Mountain whitefish	*Prosopium williamsoni*	resident
Peamouth	*Mylocheilus caurinus*	resident
Rainbow trout	*Oncorhynchus mykiss*	resident
Sockeye salmon	*Oncorhynchus nerka nerka*	migratory
Steelhead[f]	*Oncorhynchus mykiss gairdneri*	migratory
Prickly sculpin	*Cottus asper*	resident
Threespine stickleback	*Gasterosteus aculeatus*	both
Reticulate sculpin	*Cottus perplexus*	resident
Mottled sculpin	*Cottus bairdi*	resident
Paiute sculpin	*Cottus beldingi*	resident
Riffle sculpin	*Cottus gulosus*	resident
Shorthead sculpin	*Cottus confuscus*	resident
Starry flounder	*Platichthys stellatus*	migratory
Torrent sculpin	*Cottus rhotheus*	resident
Warmouth	*Lepomis gulosus*	resident
Piscivore		
Black crappie	*Pomoxis nigromaculatus*	resident
Largemouth bass	*Micropterus salmoides*	resident
Northern pikeminnow	*Ptychocheilus oregonensis*	resident

Table 4-1. Fish Known to be Present or Common in the LWR

Species	Scientific Name	Resident or Migratory[a]
Smallmouth bass	*Micropterus dolomieui*	resident
Walleye	*Stizostedion vitreum*	resident
White crappie	*Pomoxis annularis*	resident
Yellow perch	*Perca flavescens*	resident
Pacific brook lamprey	*Lampetra pacifica*	resident
Pacific lamprey[g]	*Lampetra tridentata*	migratory
River lamprey[h]	*Lampetra ayresi*	migratory
Western brook lamprey	*Lampetra richardsoni*	resident

[a] Wydoski and Whitney (2003).
[b] Known to be present in the Columbia River; federally listed as a species of concern.
[c] Federally listed as threatened in Lower Columbia River and Upper Willamette River; state listed as a critical species on ODFW sensitive species list.
[d] Federally listed as species of concern; state listed as a critical species on ODFW sensitive species list.
[e] Federally listed as threatened in Lower Columbia River; state listed as endangered on the ODFW endangered species list.
[f] Federally listed as threatened in Lower Columbia River and Upper Willamette River; state listed as a critical species on ODFW sensitive species list.
[g] Federally listed as species of concern; state listed as a vulnerable species on ODFW sensitive species list.
[h] Federally listed as species of concern.
LWR – Lower Willamette River

Table 4-2. Fish Species Known to be Rare or Possible but with Unconfirmed Presence in the LWR

Species	Scientific Name	Resident or Migratory[a]
Omnivore		
Fathead minnow	*Pimephales promelas*	resident
Black bullhead	*Ameiurus melas*	resident
Oriental weatherfish	*Misgurnus anguillicaudatus*	resident
Invertivore		
Banded killifish	*Fundulus diaphanus*	resident
Brook trout	*Salvelinus fontinalis*	migratory
Brown trout	*Salmo trutta*	resident
Lake trout	*Salvelinus namaycush*	both
Chum salmon[b]	*Oncorhynchus keta*	migratory
Kokanee	*Oncorhynchus nerka kennerlyi*	resident
Mosquitofish	*Gambusia affinis*	resident
Redside shiner	*Richardsonius balteatus*	resident
Green sunfish	*Lepomis cyanellus*	resident
Leopard dace	*Rhinichthys falcatus*	resident
Longnose dace	*Rhinichthys cataractae*	resident
Oregon chub[c]	*Oregonichthys crameri*	resident
Redear sunfish	*Lepomis microlophus*	resident
Sand roller	*Percopsis transmontana*	resident
Speckled dace	*Rhinichthys osculus*	resident
Tench	*Tinca tinca*	resident
White catfish	*Ameiurus catus*	resident
Channel catfish	*Ictalurus punctatus*	resident
Bull trout[d]	*Salvelinus confluentus*	migratory

[a] Wydoski and Whitney (2003).
[b] Federally listed as threatened in Columbia River; state listed as a critical species on ODFW sensitive species list.
[c] Federally listed as endangered.
[d] Federally listed as threatened.
LWR – Lower Willamette River

Table 4-3. Lines of Evidence Used to Evaluate Risks to Fish Receptors in the LWR

Receptor of Concern	Assessment Endpoint	Measure of Effect and Exposure (Measurement Endpoint)	Lines of Evidence in Support of Measurement Endpoint
Omnivore/Herbivore			
Largescale sucker	survival, growth, and reproduction	Water exposure concentrations compared to AWQC or water-based TRVs	Concentrations in surface water relative to reported AWQC or literature values.
		Tissue data compared to tissue-based TRVs (chemical-dependent)	Empirical whole-body concentrations relative to tissue TRVs.
			Predicted (BSAF) whole-body concentrations relative to tissue TRVs.
		Dietary dose compared to dietary TRVs (chemical-dependent)	Dietary dose compared to dietary TRVs.
		Fish condition or incidence of lesions (primarily for PAHs)	Lesion incidence compared to areas of contamination and/or lesion-based TRVs (if relevant to receptor species).[a]
Carp[b]	survival, growth, and reproduction	Tissue data compared to tissue-based TRVs (chemical-dependent)	Tissue-based TRV approach for dioxin-like contaminants using literature values and incorporating TEQs based on WHO TEFs. Risk from other compounds assessed in uncertainty analysis.
White sturgeon	survival, growth, and reproduction	Water exposure concentrations compared to AWQC or water-based TRVs	Concentrations in surface water relative to reported AWQC or literature values.
		Tissue data compared to tissue-based TRVs (chemical-dependent)	Empirical whole-body concentrations relative to tissue TRVs.
			Predicted (BSAF) whole-body concentrations relative to tissue TRVs. Specific approaches for modeling adult tissue concentrations to be developed.
		Dietary dose compared to dietary TRVs (chemical-dependent)	Dietary dose compared to dietary TRVs.
		Fish condition or incidence of lesions (primarily for PAHs)	Lesion incidence compared to areas of contamination and/or lesion-based TRVs (if relevant to receptor species).

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Table 4-3. Lines of Evidence Used to Evaluate Risks to Fish Receptors in the LWR

Receptor of Concern	Assessment Endpoint	Measure of Effect and Exposure (Measurement Endpoint)	Lines of Evidence in Support of Measurement Endpoint
Invertivore			
Sculpin	survival, growth, and reproduction	Water exposure concentrations compared to AWQC or water-based TRVs	Concentrations in surface water relative to reported AWQC or literature values.
			Concentration in TZW relative to reported AWQC or literature values.[d]
		Tissue data compared to tissue-based TRVs (chemical-dependent)	Empirical whole-body concentrations relative to tissue TRVs.
			Predicted (BSAF) whole-body concentrations relative to tissue TRVs.
		Dietary dose compared to dietary TRVs (chemical-dependent)	Dietary dose compared to dietary TRVs to also include stomach content data or other approaches refined specifically for PAHs. Existing data will be used initially (specific approach to be developed).
		Fish condition or incidence of lesions (primarily for PAHs)	Lesion incidence compared to areas of contamination and/or lesion-based TRVs (if relevant to receptor species).
Peamouth	survival, growth, and reproduction	Water exposure concentrations compared to AWQC or water-based TRVs	Concentrations in surface water relative to reported AWQC or literature values.
		Tissue data compared to tissue-based TRVs (chemical-dependent)	Empirical whole-body concentrations relative to tissue TRVs.
			Predicted (BSAF) whole-body concentrations relative to tissue TRVs.
		Dietary dose compared to dietary TRVs (chemical-dependent)	Dietary dose compared to dietary TRVs.
		Fish condition or incidence of lesions (primarily for PAHs)	Lesion incidence compared to areas of contamination and/or lesion-based TRVs (if relevant to receptor species[a]).
Juvenile chinook salmon	survival and growth (including reproduction as a surrogate for growth)	Water exposure concentrations compared to AWQC or water-based TRVs	Concentrations in surface water relative to reported AWQC or literature values.
		Tissue data compared to tissue-based TRVs (chemical-dependent)	Empirical whole-body concentrations relative to tissue TRVs.
			Predicted (BSAF) whole-body concentrations relative to tissue TRVs.
		Dietary dose compared to dietary TRVs (chemical-dependent)	Dietary dose compared to dietary TRVs, including stomach content data or other approaches refined specifically for PAHs. Existing data will be used initially (specific approach to be developed).

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Table 4-3. Lines of Evidence Used to Evaluate Risks to Fish Receptors in the LWR

Receptor of Concern	Assessment Endpoint	Measure of Effect and Exposure (Measurement Endpoint)	Lines of Evidence in Support of Measurement Endpoint
		Fish condition or incidence of lesions (primarily for PAHs)	Lesion incidence compared to areas of contamination and/or lesion-based TRVs (if relevant to receptor species).[a]
Adult chinook salmon[c]	survival, growth, and reproduction	Comparison of surface water concentrations to olfaction-based TRVs for metals	Surface water data was evaluated to determine if contaminant concentrations exceed concentrations known to be associated with changes in olfactory function that affect swimming, homing behavior and, ultimately, reproduction.
Piscivore			
Smallmouth bass	survival, growth, and reproduction	Water exposure concentrations compared to AWQC or water-based TRVs	Concentrations in surface water relative to reported AWQC or literature values.
		Tissue data compared to tissue-based TRVs (chemical-dependent)	Empirical whole-body concentrations relative to tissue TRVs.
			Predicted (BSAF) whole-body concentrations relative to tissue TRVs.
		Dietary dose compared to dietary TRVs (chemical-dependent)	Dietary dose compared to dietary TRVs.
		Fish condition or incidence of lesions (primarily for PAHs)	Lesion incidence compared to areas of contamination and/or lesion-based TRVs (if relevant to receptor species).[a]
Northern pikeminnow	survival, growth, and reproduction	Water exposure concentrations compared to AWQC or water-based TRVs	Concentrations in surface water relative to reported AWQC or literature values.
		Tissue data compared to tissue-based TRVs (chemical-dependent)	Empirical whole-body concentrations relative to tissue TRVs.
			Predicted (BSAF or FWM) whole-body concentrations relative to tissue TRVs.
		Dietary dose compared to dietary TRVs (chemical-dependent)	Dietary dose compared to dietary TRVs.

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Table 4-3. Lines of Evidence Used to Evaluate Risks to Fish Receptors in the LWR

Receptor of Concern	Assessment Endpoint	Measure of Effect and Exposure (Measurement Endpoint)	Lines of Evidence in Support of Measurement Endpoint
Detritivore			
Pacific lamprey ammocoetes	survival and growth	Tissue data compared to tissue-based TRVs (chemical-dependent)	Empirical whole-body concentrations relative to tissue TRVs or surrogate (to potentially include most sensitive of all aquatic species, not just fish).
			Predicted (BSAF) whole-body concentrations relative to tissue TRVs.
		Water exposure concentrations compared to AWQC or water-based TRVs	Water concentrations compared to literature-based or AWQC criteria for protection of early life stages.
			Concentration in transition zone water relative to reported AWQC or literature values.[d]
			Water concentrations compared to reported AWQC or literature TRVs. Relevance of TRVs to lamprey to be confirmed following acute laboratory studies with ammocoetes.

[a] Incidence of lesions was evaluated for benthic fish only.

[b] Carp is not a selected receptor of concern for the ERA; whole-body carp tissue was analyzed for dioxins and dioxin-like chemicals (including PCB congeners), was used as a surrogate for other fish species.

[c] Adult chinook salmon was not a selected ecological receptor of concern; however, this LOE was evaluated per EPA request.

AWQC – ambient water quality criteria

BSAF – biota-sediment accumulation factor

FWM – food web model

LWR – Lower Willamette River

TRV – toxicity reference value

TZW – transition zone water

WHO – World Health Organization

EPA will provide further direction on the approaches for these LOEs in future evaluations.

Lamprey – because of the timing of the data collection for these LOEs, these LOEs will be presented in the BERA.

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Table 4-4. Summary of Fish Tissue Round 2 COPCs

Round 2 COPC	Largescale Sucker	Sculpin	Peamouth	Juvenile Chinook Salmon	Smallmouth Bass	Northern Pikeminnow
Chromium	X					
Lead			X			
Mercury						X
Selenium		X[a]				
Zinc				X		
BEHP	X	X			X	
Total PCBs	X	X			X	X
4,4′-DDD	X	X		X	X	
4,4′-DDT		X				
Total DDTs	X	X			X	X
beta-HCH		X				

[a] Selenium was identified as a Round 2 COPC based on the predicted tissue screen.
BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
HCH – hexachlorocyclohexane
PCB – polychlorinated biphenyl

Table 4-5. Summary of Dietary Dose Round 2 COPCs for Fish

Round 2 COPC	Largescale Sucker	Pre-Breeding Sturgeon	Sculpin	Peamouth	Juvenile Chinook Salmon	Smallmouth Bass	Northern Pikeminnow
Cadmium	X	X	X	X	X	X	X
Copper	X	X	X	X	X	X	X
Mercury		X					X
Tributyltin ion	X	X	X	X	X	X	X
Benzo(a) pyrene		X					
Total PAHs		X					
Total PCBs	X	X	X	X	X	X	X
Total DDTs	X	X	X	X	X	X	X

COPC – chemical of potential concern
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl

Table 4-6. Summary of Surface Water Round 2 COPCs for Fish

Round 2 COPCs	
Metals	
Zinc (dissolved)	
PAHs	
Benzo(a)anthracene	Benzo(a)pyrene
SVOCs	
4-Chloro-3-methylphenol	
PCBs	
Total PCBs	
Pesticides	
2,4′-DDD	4,4′-DDT
2,4′-DDT	Total DDTs
4,4′-DDD	

COPC – chemical of potential concern
PAH – polycyclic aromatic hydrocarbons
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound

Table 4-7. Round 2 COPCs Identified in TZW for Fish

Round 2 COPCs	
Metals	
Barium (total)[a]	Nickel (dissolved)[b]
Cadmium (dissolved)[b]	Silver (total)[b]
Chromium (total)[b]	Sodium (total)[a]
Copper (dissolved)[b]	Vanadium (total)[a]
Lead (dissolved)[b]	Zinc (dissolved)[b]
PAHs	
2-Methylnaphthalene	Chrysene
Acenaphthene	Dibenzo(a,h)anthracene
Anthracene	Fluoranthene
Benzo(a)anthracene	Fluorene
Benzo(a)pyrene	Indeno(1,2,3-cd)pyrene
Benzo(b)fluoranthene	Naphthalene
Benzo(g,h,i)perylene	Phenanthrene
Benzo(k)fluoranthene	Pyrene
SVOCs	
1,2-Dichlorobenzene	Dibenzofuran
1,4-Dichlorobenzene	
Pesticides	
2,4′-DDD	4,4′-DDE
2,4′-DDT	4,4′-DDT
4,4′-DDD	Total DDTs
Herbicides	
Dalapon	Silvex
VOC	
1,1-Dichloroethene	Ethylbenzene
1,2,4-Trimethylbenzene	Isopropylbenzene
1,3,5-Trimethylbenzene	m,p-Xylene
Benzene	o-Xylene
Carbon disulfide	Toluene
Chlorobenzene	Total xylenes
Chloroethane	Trichloroethene
cis-1,2-Dichloroethene	Vinyl chloride
Cyanide	
Cyanide	
Perchlorate	
Perchlorate	

[a] Eco SL is based on a total concentration; dissolved concentrations were not evaluated.
[b] Eco SL is based on a dissolved concentration; total concentrations were not evaluated.

COPC – chemical of potential concern
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound
TZW – transition zone water
VOC – volatile organic compound

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Table 4-8. Largescale Sucker Tissue EPCs

Sample ID	Tissue EPCs				
	Chromium (mg/kg)	**BEHP (µg/kg)**	**Total PCBs (µg/kg)**	**4,4′-DDD (µg/kg)**	**Total DDTs (µg/kg)**
03R014	0.78	78 UT	2,060 J	41 T	228 J
03R014 (FR)	0.38	99 UT	358	31 J	176 NJ
05R006	0.60	97 UT	96.9	20	152 NJ
07R009	0.41	78 UT	1,650	57 T	196 J
08R010	2.77 T	3,000 JT	525	150 T	673
09R006	1.76	800 T	738	27.5 JT	155 J
UCL (all samples)	**1.9**	**5,500[a]**	**1,540**	**110**	**624**

[a] UCL exceeds the maximum concentration.
BEHP – bis(2-ethylhexyl) phthalate
EPC – exposure point concentration
FR – field replicate
ID – identification
J – estimated concentration
N – presumptive evidence of a compound
PCB – polychlorinated biphenyl
T – value calculated or selected from multiple results
U – not detected at given concentration
UCL – upper confidence limit

Table 4-9. Sculpin Tissue EPCs

Sample ID	Tissue EPCs					
	BEHP (µg/kg)	**Total PCBs (µg/kg)**	**4,4′-DDD (µg/kg)**	**4,4′-DDT (µg/kg)**	**Total DDTs (µg/kg)**	**beta-HCH (µg/kg)**
02R001	92 UT	2,370 J	14 JT	6.3 UT	56.4 J	4 UT
02R015	400 UT	3,400	9.2 JT	6.3 UT	51.6 J	8.5 UT
03R001	150 UT	146 J	5.2 NJ	16 J	40.6 NJ	4.3 NJ
03R002	180 UT	123 J	6.7 N	19	45.9 NJ	4.1 NJ
03R002 (FR)	96 UT	172 J	7.2 N	37	72.2 NJ	6.2 NJ
03R004	810 UT	334	5.7 N	54	102 N	4.5 NJ
03R004 (FR)	140 UT	315	8.3 N	50	110 NJ	2.9 NJ
03R005	160 UT	370	7.7 N	75	144 NJ	1 UJ
03R032	86 UT	176	7.3 N	23 J	57.1 NJ	4.1 NJ
03R034	150 UT	538	6.1 UT	12 JT	46.5 J	4.8 UT
04R002	240 UT	166 J	5.6 J	26 J	63.2 NJ	3.8 NJ
04R003	260 UT	198 J	5.3 N	21 J	51.3 NJ	2.5 NJ
04R004	9,400 JT	125 J	12 J	26 J	77.1 NJ	3.4 NJ
05R001	96 UT	198 J	6.1 UT	6.3 UT	31.7 J	8.5 UT
05R020	170 UT	134 J	11 J	22 J	50.3 J	3.9 NJ
06R001	82 UT	100 J	16 J	27 J	82.8 NJ	5.3 NJ
06R002	99 T	691 J	6.1 UT	6.3 UT	32.7 J	8.5 UT
06R002 (FR)	260 UT	2,450	12 JT	11 JT	60.5 J	8.5 UT
06R004	200 UT	588 J	140 JT	135 JT	570 J	8.5 UT
07R003	100 UT	236 J	84 T	250 T	582 J	8.5 UT
07R006	220 UT	440 J	305 T	1,700 T	3,060	1 UJT
08R001	160 UT	189	6.1 UT	8.9 JT	52.5 J	8.5 UT
08R002	100 UT	159	4.6 NJ	21 J	45.7 NJ	2 NJ
08R003	28,000 JT	583 J	6.1 UT	6.3 UT	31.7 J	4 UT
09R001	220 UT	651 J	4 UT	19 JT	52.5 J	8.5 UT
09R002	120 UT	823	9.6 U	81	182 NJ	9.6 U
UCL (all samples)	**13,000**	**1,300**	**150**	**750**	**1,400**	**4.8**

BEHP – bis(2-ethylhexyl) phthalate
EPC – exposure point concentration
FR – field replicate
HCH – hexachlorocyclohexane
ID – identification
J – estimated concentration
N – presumptive evidence of a compound
T – value calculated or selected from multiple results
U – not detected at given concentration
UCL – upper confidence limit

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Table 4-10. Peamouth Tissue EPCs

Sample ID	Tissue EPCs for Lead (mg/kg)
03R014	10.6
05R006	0.111
08R010	0.047
09R006	0.031
UCL (all samples)	**37**[a]

[a] UCL exceeds maximum concentration.
EPC – exposure point concentration
ID – identification
UCL – upper confidence limit

Table 4-11. Juvenile Chinook Salmon Tissue EPCs

Sample ID	Tissue EPCs	
	Zinc (mg/kg)	4,4′-DDD (µg/kg)
T01-REP1	27.9 T	1.0 J
T01-REP2	25.935	1.9
T01-REP3	27.676	1.6
T02-REP1	30.81	110
T02-REP2	29.008	120
T02-REP3	29.1	130
T03-REP1	24.038	1.2 J
T03-REP2	31.52	1.5 J
T03-REP3	28.665	1.4 J
02R102	31.1	4.6 N
02R112	33.3	6 JT
02R113	33.3	5.2 JT
03R118	29.6	4.2 J
03R125	29.6	6.3 J
04R126	31.5	8.1 J
UCL (all samples)	**30.7**	**150**[a]

[a] UCL exceeds maximum concentration.
EPC – exposure point concentration
ID – identification
J – estimated concentration
N – presumptive evidence of a compound
T – value calculated or selected from multiple results
UCL – upper confidence limit

Table 4-12. Smallmouth Bass Tissue EPCs

Sample ID	Tissue EPCs			
	BEHP (µg/kg)	**Total PCBs (µg/kg)**	**4,4′-DDD (µg/kg)**	**Total DDTs (µg/kg)**
03R014	1,300 UT	788	30.5 JT	199 J
04R023	32,000 JT	1,320 J	47 T	309 J
04R023 (FR)	690 UT	478	33.5 JT	185 J
04R023 (FR)	87,000 JT	600 J	31 JT	140 J
05R006	92 UT	400 J	35 JT	185 J
06R024	96 UT	278 J	19 JT	149 J
07R009	86 UT	544	58 T	416
07R009 (FR)	630 UT	109 J	55 T	365 J
07R009 (FR)	98 UT	818 J	110 T	341
08R010	99 UT	4,950 J	11 JT	122 J
08R010 (FR)	96 UT	3,610 J	12 JT	79.7 J
08R010 (FR)	94 UT	1,100	25.5 T	122 J
08R032	96 UT	918 J	25 JT	189 J
09R006	96 UT	878 J	37.5 T	192
UCL (all samples)	**73,000**	**1,960**	**51**	**262**

BEHP – bis(2-ethylhexyl) phthalate
EPC – exposure point concentration
FR – field replicate
ID – identification
J – estimated concentration
T – value calculated or selected from multiple results
U – not detected at given concentration
UCL – upper confidence limit

Table 4-13. Northern Pikeminnow Tissue EPCs

Sample ID	Tissue EPCs		
	Mercury (µg/kg)	**Total PCBs (µg/kg)**	**Total DDTs (µg/kg)**
03R014	0.173	729	286
03R014 (FR)	0.146 T	378	261
05R006	0.326	448	365
07R009	0.494	1,930	764
08R010	0.153	795	172 NJ
09R006	0.39	1,030	163
UCL (all samples)	**0.40**	**1,350**	**593**

EPC – exposure point concentration
FR – field replicate
ID – identification
J – estimated concentration
N – presumptive evidence of a compound
T – value calculated or selected from multiple results
UCL – upper confidence limit
FR –field replicate

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Table 4-14. Chromium Tissue-Residue Toxicity Studies for Fish

Chemical Form	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Conditions	Endpoint Effects	Source
Hexavalent chromium	rainbow trout (fingerlings)	0.9 – **5.5**[a]	**8.7** – 10.5[b]	aqueous exposure for 2 to 4 days	50% survival (LC50)	Van der Putte et al. (1981)

[a] 10.5 is LC50 at pH 7.8; 25% mortality was observed in fish with residues of 8.7 mg/kg ww at pH 6.5; study was uncontrolled.

[b] 100% survival was reported in fish exposed to 2.0 to 5.0 mg/L at pH 6.5 and 7.8; study was uncontrolled.

LC50 – concentration that is lethal to 50% of an exposed population

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as TRVs.

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Table 4-15. Lead Tissue-Residue Toxicity Studies for Fish

Chemical Form	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Conditions	Endpoint Effects	Source
Lead nitrate	brown trout yearlings		0.10[a]	single IP injection	reduced body weight and immune activity	O'Neil (1981)
Lead nitrate	rainbow trout fry	2.0, 2.58[b]		aqueous and dietary exposure for 60 days[c]	100% survival, no effect on body weight or length	Mount et al. (1994)
Lead nitrate	brown trout yearlings		3.0[a]	single IP injection	0% survival	O'Neil (1981)
Lead nitrate	brook trout egg – 12 weeks	4.02[b]		aqueous exposure for three generations	no effect on survival	Holcombe et al. (1976)
Lead nitrate	brook trout egg – 12 weeks	**2.54**[b]	**4.02**[b]	aqueous exposure for three generations	reduced body weight	Holcombe et al. (1976)
Lead acetate	juvenile fathead minnow		26.2	aqueous exposure for 4 weeks	behavior (altered feeding behavior and ability to capture prey)	Weber et al. (1991)
Lead nitrate	largemouth bass fingerlings		10 – 45[c]	aqueous exposure for 96 hours	50% survival (LC50)	Coello and Khan (1996)
Lead nitrate	goldfish fingerlings		92 – 315[c]	aqueous exposure for 96 hours	50% survival (LC50)	Coello and Khan (1996)
Lead nitrate	green sunfish fingerlings		275 – 625[c]	aqueous exposure for 96 hours	50% survival (LC50)	Coello and Khan (1996)

[a] LOAEL is injected dose
[b] Tissue concentration TRVs were converted into to wet weight from dry weight assuming 80% moisture content.
[c] LOAEL TRVs are the range of whole body lead concentrations in fish.
IP – intraperitoneal
LC50 – concentration that is lethal to 50% of an exposed population
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
ww – wet weight
Bold identifies the NOAEL and LOAEL selected as TRVs.

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Table 4-16. Mercury Tissue-Residue Toxicity Studies for Fish

Chemical Form	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Conditions	Endpoint Effects	Source
Mercuric chloride	guppy male adults	0.2		sediment and aqueous for 20 days	no effect on survival	Kudo and Mortimer (1979)
Methyl-mercury	golden shiner		0.23	dietary for 90 days	altered predator avoidance	Webber and Haines (2003)
Methyl-mercury	walleye juvenile		0.254[a]	dietary for 6 months	reduced male growth	Friedmann et al. (1996)
Methyl-mercuric chloride	mummichog adults	**0.2**	**0.47**	aqueous for 42 days	reduced male survival	Matta et al. (2001)
Methyl-mercury	fathead minnow		0.56[b]	dietary for multiple generation	46% spawning success	Hammerschmidt et al. (2002)
Inorganic mercury	catfish (sac-fry converted to adult whole body)		0.76[c] (egg = 0.014)	aqueous, from spawning until 4 days post-hatch	67% survival	Birge et al. (1979)
Mercuric chloride	fathead minnow 3 months	0.8	1.31	dietary for 60 days	reduced growth	Snarski and Olson (1982)
Methyl-mercuric chloride	rainbow trout fingerling	2.28		aqueous for 24 days	no effect on growth	Phillips and Buhler (1978)
Inorganic mercury	rainbow trout alevins (sac-fry converted to adult whole body)		2.3[d]	exposed to contaminated sediment for 20 days	egg/embryo survival reduced by 46%	Birge et al. (1977)
Chemical form of mercury unknown	rainbow trout 4 days post-hatch (sac-fry converted to adult whole body)		2.6[e]	aqueous, from spawning until 4 days post-hatch	45% survival	Birge et al. (1979)
Methyl-mercury	brook trout embryo (adult)	2.7	3.4	aqueous for 756 days	reduced number of viable eggs produced	McKim et al. (1976)
Mercuric chloride	fathead minnow (3 months old)	2.75	4.18	aqueous for 60 days	reduced survival	Snarski and Olson (1982)
Mercuric chloride	fathead minnow larvae-adult	2.84	4.47	aqueous for 287 days	altered reproductive success	Snarski and Olson (1982)
Methyl-mercury	rainbow trout juvenile	5		aqueous for 84 days	no effect on growth or survival	Lock (1975)
Mercuric chloride	brown trout		5	aqueous for 5 days	altered behavior (mercury deposition in inner ear)	Skak and Baatrup (1993)
Mercuric chloride	goldfish		5.6[b]	aqueous for 2 days	reduced survival	Heisinger et al. (1975)
Methyl-mercuric chloride	rainbow trout fingerling	5.67		aqueous for 24 days	no effect on growth	Phillips and Buhler (1978)

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Table 4-16. Mercury Tissue-Residue Toxicity Studies for Fish

Chemical Form	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Conditions	Endpoint Effects	Source
Methyl-mercuric chloride	bluegill juvenile		6.5	aqueous for 12.5 days	reduced survival	Cember et al. (1978)
Methyl-mercuric chloride	rainbow trout fingerling	8.63		aqueous for 24 days	no effect on growth	Phillips and Buhler (1978)
Methyl-mercuric chloride	brook trout (39 weeks old)	9.4		aqueous for 756 days	no effect on juvenile weight	McKim et al. (1976)
Methyl-mercury	rainbow trout fingerling		10	dietary for 84 days	reduced growth and final body weight	Rodgers and Beamish (1982)
Methyl-mercury	fathead minnow	10.9		aqueous for 336 days	no effect on growth, survival or behavior	Olson et al. (Olson et al. 1975)
Methyl-mercuric chloride	mummichog	1.1	11	dietary for 42 days	reduced F1 fertilization success	Matta et al. (2001)
Methyl-mercuric chloride	brown trout		11.4	gastric gavage for 4 weeks	altered behavior (mercury deposition in inner ear)	Skak and Baatrup (1993)
Methyl-mercuric chloride	rainbow trout sub-adult	12		aqueous for 75 days	no effect on body weight or survival	Niimi and Lowe-Jinde (1984)
Methyl-mercuric chloride	mummichog	12		dietary for 42 days	no effect on F1 hatchability, survival, fecundity, F2 larval survival	Matta et al. (2001)
Methyl-mercury	grayling sac-fry (converted to adult whole body)	5.80 (egg = 0.09)	17.4[f] (egg = 0.27)	aqueous for first 10 days of development	reduced feeding efficiency, reduced competitive ability	Fjeld et al. (1998)
Methyl-mercury	rainbow trout fingerling	29		84 days	no effect on survival	Rodgers and Beamish (1982)
Methyl-mercury	walleye (field-collected egg converted to whole body)[i]	35.69[g] (egg = 0.99)		maternal transfer	no effect on fertilization and hatching success, larval length	Latif et al. (2001)
Methyl-mercury	grayling sac-fry (converted to adult whole body)		244.7[f] (egg = 3.8)	aqueous for first 10 days of development	reduced fry survival, incidence of jaw deformities	Fjeld et al. (1998)
Methyl-mercury	Japanese medaka (egg converted to adult whole body)[k]	577 (egg = 16)	1,045[h] (egg = 29)	aqueous for 16 days	altered offspring behavior (equilibrium loss) and hatchability	Heisinger and Green (1975)

[a] LOAEL is whole-body concentration minus viscera; growth effect was not dose-responsive.

[b] LOAEL was converted to ww assuming 80% moisture in whole fish.

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[c] Adult concentration was estimated using sac-fry:adult conversion factor of 54.1 based on average data reported in five fish species in Niimi (1983) and assuming 3:2 ratio of concentration of sac-fry:egg (see Appendix B of the Ecological PRE (Windward 2005) for details on use and derivation of conversion factors).

[d] LOAEL is based on alevin (sac-fry) concentration of 0.036. Adult concentration was estimated using sac-fry:adult conversion factor of 64.4 based on rainbow trout data from Niimi (1983) and assuming 3:2 ratio of concentration of sac-fry:egg (see Appendix B of the Ecological PRE (Windward 2005)for details on use and derivation of conversion factors).

[e] LOAEL is sac-fry concentration of 0.0682 µg/g ww. Adult concentration was estimated using sac-fry:adult conversion factor of 64.4 based on rainbow trout data from Niimi (1983) and assuming 3:2 ratio of concentration of sac-fry:egg (see Appendix B of the Ecological PRE (Windward 2005) for details on use and derivation of conversion factors).

[f] NOAEL and LOAEL were reported as yolk-fry concentrations. Adult tissue concentration was estimated using sac-fry:adult conversion factor of 64.4 based on rainbow trout data from Niimi (1983) and assuming 3:2 ratio of concentration of sac-fry:egg (see Appendix B of the Ecological PRE (Windward 2005) for details on use and derivation of conversion factors).

[g] Study used field-collected fish. Adult concentration was estimated using egg:adult conversion factor of 36.05 based on average data reported in five species in Niimi (1983) (see Appendix B of the Ecological PRE (Windward 2005) for details on use and derivation of conversion factors).

[h] Adult concentration was estimated using egg:adult conversion factor of 36.05 based on average data reported in five species in Niimi (1983) (see Appendix B of the Ecological PRE (Windward 2005) for details on use and derivation of conversion factors).

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as TRVs.

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Table 4-17. Zinc Tissue-Residue Toxicity Studies for Fish

Chemical Form	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Conditions	Endpoint Effects	Source
Zinc proteinate	rainbow trout	17.5[a]		dietary exposure for 84 days	no effect on growth	Hardy and Shearer (1985)
Zinc sulfate	brook trout sac-fry		22.6[b]	aqueous exposure for 24 weeks	reduced hatchability of second-generation embryos	Holcombe et al. (1979)
Zinc chloride	striped mullet fry		23.1 – 33.3[c]	aqueous exposure for up to 72 hours	0 to 50% survival	Zyadah and Abdel-Baky (2000)
Zinc sulfate	rainbow trout fingerlings	35.8[a]		aqueous and dietary exposure for 16 weeks	no effect on growth or survival[d]	Spry (1988)
Zinc sulfate	flagfish juveniles	**34**[a]	**40**	aqueous exposure for two generations	reduced female body weight	Spehar (1976)
Zinc sulfate	Atlantic salmon juvenile	60		aqueous exposure for 80 days	no effect on growth or survival	Farmer et al. (1979)
Zinc sulfate	flagfish juveniles	44[a]	60[a]	aqueous exposure for two generations	reduced male growth and mean number of spawnings	Spehar (1976)
Zinc chloride	rainbow trout fry	78.4[a, e]		aqueous and dietary exposure for up to 60 days	100% survival, no effect on body weight and length	Mount et al. (1994)
Zinc sulfate	guppies fry (5 days old)	41.4[a]	88.8	aqueous exposure for 134 days	delayed onset of sexual maturity	Pierson (1981)
Zinc sulfate	guppies fry (5 days old)		297[a]	aqueous exposure for 135 days	reduced female body weight and number of females giving birth	Pierson (1981)
Zinc sulfate	rainbow trout		5,500	aqueous exposure	reduced survival (estimated LC50)	Connolly (1985); Brown et al. (1969)

[a] Converted from dry weight assuming 80% moisture content.
[b] Concentration in newly hatched larvae (sac-fry) following exposure of parents; no egg to adult conversion factor available to estimate adult tissue concentration.
[c] Range of whole-body residues in fish exposed to 30 to 60 mg/L. Control mortality was not presented.
[d] Adverse effects resulting from zinc deficiency were observed at the LOAEL.
[e] Fish were exposed to dietary zinc and to 50 ppb zinc and other metals in water.

LC50 – concentration that is lethal to 50% of an exposed population
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
Bold identifies the NOAEL and LOAEL selected as TRVs.

Table 4-18. BEHP Tissue-Residue Toxicity Studies for Fish

Chemical Form	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Conditions	Endpoint Effects	Source
BEHP	rainbow trout	0.18 – 1.03		aqueous exposure for up to 96 hours	no effect on metabolic clearance rate	Tarr et al. (1990)
BEHP	rainbow trout (sac-fry)	**0.39**	**10.6[a] (egg = 1.5)**	aqueous exposure 10 days prior to hatching	reduced sac-fry survival	Mehrle and Mayer (1976)
BEHP	sheepshead minnows	1.0 - 6.75		aqueous exposure for 24 hours	no effect on survival	Wofford et al. (1981)

[a] Fry tissue residues were calculated from bioconcentration factor and water concentrations reported in paper (tissue concentration = [BCF x water concentration]/1,000); the LOAEL was based on tissue residues measured in surviving fry 20 days after mortality was observed in sac-fry; adult concentration was estimated using sac-fry: adult conversion factor of 7.02 based on Niimi (1983). See Appendix B of the Ecological PRE (Windward 2005) for detail on use and derivation of conversion factors).

BEHP – bis(2-ethylhexyl) phthalate

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as TRVs.

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Table 4-19. PCBs Tissue-Residue Toxicity Studies for Fish

Chemical Form	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Conditions	Endpoint Effects	Source
Aroclor 1254	coho salmon (yearlings)		0.15[a]	injection (single)	94% survival; reduction in ability to adapt to seawater	Folmar et al. (1982)
Aroclor 1260	barbel (adult)		0.52[b,c]	dietary exposure for 50 days	reduced fecundity	Hugla and Thome (1999)
Aroclor 1254	chinook salmon (juvenile)	0.98		dietary exposure for 4 weeks	no effect on growth or survival	Powell et al. (2003)
Aroclor 1254	brook trout (fry)		1.53[d]	aqueous and dietary exposure for 176 days	reduced fry survival	Berlin et al. (1981)
PCB mixture	lake trout fry	1.8[b]		aqueous and dietary exposure for approx 52 days	increase in growth and length	Mac and Seelye (1981b)
Aroclor 1242, 1254, or 1260	fathead minnow (6 months)		1.86 – 749[e]	aqueous exposure for up to 300 hours	lethal body burden (reduced survival)	van Wezel et al. (1995)
Aroclor 1260	barbel (adult)		2.6[b,c]	dietary exposure for 50 days	failure to spawn	Hugla and Thome (1999)
Mixture of PCB congeners (PCB 77, 126, 118, 105, 167, 156, 159, 189)	mummichog (spawning females)	0.76[a]	3.8[a]	IP injection; for 50 days during spawning	reduced female survival, and egg production	Black et al. (1998)
Aroclor mixture	Atlantic salmon egg (converted to adult whole body)		**4.02[f] (egg = 0.857)**	aqueous exposure for 48 hours	reduced live fry body weight	Fisher et al. (1994)
Aroclor 1254	rainbow trout egg (converted to adult whole body)		7.69[f] (egg = 1.64)	maternal transfer for 60 days	reduced fry growth	Hendricks et al. (1981)
Aroclor 1254	rainbow trout (14 weeks)	8		dietary exposure for 32 weeks	no effect on growth or survival	Lieb et al. (1974)
Aroclor 1254	Atlantic croaker egg (converted to adult whole body)		8.67[f,g] (egg = 3.2)	maternal transfer to eggs, 2 weeks during reproduction (adults)	reduction in larval growth rate and impaired response to startle stimulus	McCarthy et al. (2003)
Aroclor 1254	sheepshead minnow adult	**1.9**	9.3	exposure for 28 days	decreased fry survival	Hansen et al. (1973)
Aroclor mixture	lake trout egg (converted to adult whole body)		12.6[f] (egg = 2.69)	parental transfer	decreased survivorship of fry	Mac et al. (1985)
Aroclor 1242	channel catfish fingerling		14[h]	dietary exposure for 20 weeks	reduced weight gain [60% of controls])	Hansen et al. (1976)

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Table 4-19. PCBs Tissue-Residue Toxicity Studies for Fish

Chemical Form	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Conditions	Endpoint Effects	Source
Aroclor 1268	mummichog adult	15		dietary exposure for approximately 6 weeks	no effect on fertilization and hatching success, or larval survival	Matta et al. (2001)
Aroclor 1254	pinfish juvenile	17		aqueous exposure for 48 hours	no effect on survival	Duke et al. (1970)
Aroclor mixture	Atlantic salmon egg		26.2 (egg = 5.59)[f]	aqueous exposure for 48 hours	retarded phototropism behavior in alevins	Fisher et al. (1994)
Aroclor 1254	lake trout sac-fry	21.1[i] (egg = 3)	31.7[i] (egg = 4.5)	aqueous and dietary exposure for 48 days	reduced fry survival	Mac and Seelye (1981a)
Aroclor 1260	channel catfish fingerling	32		dietary exposure for 193 days	no effect on growth or survival	Mayer et al. (1977)
Aroclor 1254	spot	27	46	aqueous exposure for 20 days	reduced survival	Hansen et al. (1971)
Aroclor 1254	chinook salmon juvenile	60		oral gavage, 96 hours	no effect on survival	Powell et al. (2003)
Aroclor 1254	brook trout fry (exposure to eggs)	31	71[k]	aqueous exposure for 10 days prior to hatch and 118 days after hatch	reduced growth	Mauck et al. (1978)
Aroclor 1016	sheepshead minnow fry	77		aqueous exposure for 2 weeks	no effect on fertilization and hatching success, or larval survival	Hansen et al. (1975)
Aroclor 1016	pinfish		106	aqueous exposure for 33 days	reduced survival and altered behavior (loss of equilibrium; erratic swimming)	Hansen et al. (1974)
Aroclor 1254 and 1260	rainbow trout young	120		aqueous exposure for 90 days	no effect on survival	Mayer et al. (1985)
Aroclor 1254 and 1260	rainbow trout young	70	120	aqueous exposure for 90 days	reduced growth	Mayer et al. (1985)
Aroclor 1254	brook trout fry (exposure to eggs)	71	125	aqueous exposure for 10 days prior to hatch and 118 days after hatch	reduced fry survival	Mauck et al. (1978)
Aroclor 1016	sheepshead minnow fry	77	200	aqueous exposure for 28 days	reduced fry survival	Hansen et al. (1975)
PCBs mixture	goldfish		250	aqueous exposure for 5 to 21 days	reduced survival	Hattula and Karlog (1972)

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Table 4-19. PCBs Tissue-Residue Toxicity Studies for Fish

Chemical Form	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Conditions	Endpoint Effects	Source
Aroclor 1254	brook trout egg (converted to adult whole body)		365[f] (egg = 77.9)	aqueous exposure for 21 days	reduced hatchability	Freeman and Idler (1975)
Aroclor 1254	fathead minnow		458, 361	aqueous exposure for 8.5 months	reduced spawning	Nebeker et al. (1974)
Aroclor 1254	coho salmon fingerling		645	exposure for approximately 260 days	reduced survival	Mayer et al. (1977)

[a] Injected dose (whole-body residue not reported).

[b] Converted from dry weight assuming 80% moisture content.

[c] Effects observed 1 year after tissue residue was measured. Liver tissue was not included in whole-body residue.

[d] LOAEL is based on field-collected eggs from Lake Michigan with starting egg residues of 7.6 µg/g PCBs and 4.7 µg/g DDE; mortality was estimated.

[e] Range of lethal body burdens in individual fish.

[f] Adult concentration was estimated using egg:adult conversion factor of 4.69 based on rainbow trout data in Niimi (1983) (see Appendix B of the Ecological PRE (Windward 2005) for details on use and derivation of conversion factors).

[g] LOAEL is based on field-collected eggs from the Great Lakes; total DDE concentration in the eggs was 1.27 µg/g ww.

[h] Whole-body concentration minus stomach and contents.

[i] LOAEL is based on field-collected eggs from Saugatuck, Michigan, with unknown organics; LOAEL is residue at 48 days and NOAEL is control residue; adult concentration was estimated using sac fry:adult conversion factor of 7.04 based on rainbow trout data in Niimi (1983) and assuming 3:2 ratio of concentration of sac-fry:egg (see Appendix B of the Ecological PRE (Windward 2005) for details on use and derivation of conversion factors).

[j] LOAEL is residue measured at 118 days; growth effect reported at 48 days but disappeared at 118 days.

IP – intraperitoneal

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as TRVs.

Table 4-20. Lindane Tissue-Residue Toxicity Studies for Fish

Chemical Form	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Conditions	Endpoint Effects	Source
Lindane	goldfish	2.3[a]		aqueous exposure for 19 hours	no toxicity symptoms	Gakstatter and Weiss (1967)
Lindane	bluegill	1.5[a]		aqueous exposure for 19 hours	no toxicity symptoms	Gakstatter and Weiss (1967)
Lindane	fathead minnow	6.13[b]	9.53[b]	aqueous exposure for 48 days	reduced survival	Macek et al. (1976)
Lindane	sheepshead minnow	**1.58**[c]	**79**	aqueous exposure for 96 hours	50% survival (LC50)	Schimmel et al. (1977)
Lindane	gudgeon	0.187 (brain); 0.149 (liver); 0.013 (muscle)	3.527 (brain); 5.005 (liver); 1.068 (muscle)	aqueous exposure for 96 hours	50% survival (LC50)	Marcelle and Thome (1983)

[a] Whole-body concentration excludes brain tissue (accounting for < 2% of total body insecticide count).
[b] Residues were measured in eviscerated carcasses.
[c] NOAEL extrapolated from LC50 using a UF of 50.
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
ww – wet weight
Bold identifies the NOAEL and LOAEL selected as TRVs.

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Table 4-21. DDT Tissue-Residue Toxicity Studies for Fish

Chemical Form	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Conditions	Endpoint Effects	Source
DDT; total DDE	lake trout egg		7.33[a,b] (egg = 1.27)	parental transfer	decreased fry survivorship	Mac et al. (1985)
Total DDT	rainbow trout juvenile	1.07		dietary exposure	no effect on final body weight or survival, and no abnormal behavior	Hilton et al. (1983)
Total DDT	rainbow trout fry	0.15	1.27[c,d]	field exposure	reduced early life-stage survival	Hopkins et al. (1969)
Sum of DDT, DDE, and DDD	goldfish		1.49 – 1.84	aqueous exposure for 4 days	behavior (locomotor activity reduced)	Davy et al. (1972)
DDT mixture	cutthroat trout (21 months)	**1.8[e]**	**1.8[e]**	aqueous exposure for 21 months	reduced survival	Allison et al. (1964)
sum of DDT, DDE, and DDD	brook trout juvenile	1.92		120 days	no effect on survival	Macek and Korn (1970)
DDT mixture	cutthroat trout yearlings	1.2	2.0[f]	aqueous exposure for 20 months	reduced survival (mortality observed at 4 months)	Allison et al. (1963)
Total DDT	speckled, rainbow, and cutthroat trout egg	1.03[a,g] (egg = 0.178)	2.68[a,g] (egg = 0.464)	field exposure	reduced fry survival	Cuerrier et al. (1967)
Sum of DDD, DDE, and DDT	brook trout (sexually mature yearlings)		2.8 – 3.0	dietary exposure for 156 days	reduced offspring (sac-fry and embryo) survival	Macek (1968b)
Sum of DDE and DDD	chinook salmon	0.62	3.65	dietary exposure for 40 days	reduced survival	Buhler et al. (1969)
"Total DDT"	bluegill		4.2[h]	aqueous exposure for 5 hours (32-day recovery)	altered behavior (equilibrium loss and convulsions)	Gakstatter and Weiss (1967)
Sum of DDT, DDE, and DDD	rainbow trout	4.67		dietary exposure for 140 days	no effect on growth or survival	Macek et al. (1970)
"Total DDT"	goldfish		5.1[h]	aqueous exposure for 6 hours (32 day recovery)	altered behavior (equilibrium loss and convulsions)	Gakstatter and Weiss (1967)
Total DDT	killifish		5.2	aqueous exposure for 24 hours	75% survival at 24 hours	Crawford and Guarino (1976)
Sum of DDD, DDE, and DDT	brook trout fry	7.6		dietary exposure for 156 days	no effect on growth or survival	Macek (1968b)
Sum of DDD, DDE, and DDT	brook trout under yearling	11.2[i]		dietary exposure for 31 weeks	increased growth	Macek (1968a)
Sum of DDT, DDE, and DDD	chinook salmon	11.4	12.1	dietary exposure for 40 days	reduced survival	Buhler et al. (1969)

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Table 4-21. DDT Tissue-Residue Toxicity Studies for Fish

Chemical Form	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Conditions	Endpoint Effects	Source
Sum of DDD, DDE, and DDT	brook trout under yearling		20.2 – 45.8	dietary exposure for 26 weeks	reduced survival during stress [starvation]	Macek (1968a)
Sum of DDT, DDE, and DDD	green sunfish/ pumpkinseed		24	aqueous exposure for 90 days	reduced survival	Hamelink et al. (1971)
Sum of DDT, DDE, and DDD	mosquito fish		26.5	aqueous exposure for 16 days	reduced survival	Pillai et al. (1977)
Sum of DDT, DDE, and DDD	fathead minnow juvenile-adult	40		266 days	no effect on survival	Jarvinen et al. (1976; 1977)
Sum of DDT, DDE, and DDD	sailfin molly	51.4	92.7	aqueous exposure for 21 days	reduced growth and survival	Benton et al. (1994)
Sum of DDT, DDE, and DDD	coho salmon	16.6	69.6	dietary exposure for 60 days	reduced survival	Buhler et al. (1969)
Sum of DDT, DDE, and DDD	goldfish	130[c]	200[c]	aqueous and dietary exposure for 58 days	reduced survival	Rhead and Perkins (1984)

[a] Adult concentration was estimated using egg:adult conversion factor of 5.77 based on rainbow trout data in Niimi (1983) (see Appendix B of the Ecological PRE (Windward 2005) for details on the use and derivation of conversion factors).

[b] LOAEL based on field-collected eggs from Great Lakes with PCB concentrations of 2.69 µg/g ww.

[c] Converted from dry weight to wet weight assuming 80% moisture content.

[d] LOAEL based on field-collected diet from Great Lakes; other detected chemicals in fish tissue included PCBs, mirex, dieldrin, and heptachlor epoxide.

[e] LOAEL is the tissue concentration at 111 days (3.7 months), at which mortality was significant after approximately 120 days, the tissue concentrations at this dose increased to 3.0 mg/kg at the next sampling (166 days, 5.5 months), so the actual LOAEL tissue concentration is likely somewhat higher than 1.8 mg/kg; the NOAEL is the highest tissue concentration (at 466 days) at which significant mortality was not observed over entire exposure duration of 612 days.

[f] LOAEL is tissue concentration at 56 days (1.9 months). Tissue concentrations at this dose increased to 4.3 mg/kg at the next sampling (166 days, 5.5 months), so the actual tissue concentration associated with the LOAEL is likely somewhat higher than 2.0 mg/kg; NOAEL is highest tissue concentration (at 391 days) in fish at which significant mortality was not observed over entire exposure duration.

[g] LOAEL based on field-collected eggs from wildlife stocks, hatchery, domestic stocks, commercial suppliers, and provincial and federal fish agencies.

[h] LOAEL estimated from figure presented in study; whole-body concentration excludes brain tissue (accounting for < 2% of total body insecticide count).

[i] NOAEL effect is an increase in growth (significant) and not shown to be an adverse effect.

LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
ww – wet weight
Bold identifies the NOAEL and LOAEL selected as TRVs.

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Table 4-22. Exposure Parameters Used for Fish Dietary Risk Calculations

Receptor	BW (kg)	FIR (kg ww/ day)	Moisture in Diet (%)	SI (%)[a]	SIR (kg dw/day)[a]
Largescale sucker	0.794	0.0332	85	8	0.00398
White sturgeon (pre-breeding)	15.9[b]	0.663[b]	85	8, 56[c]	0.00796; 0.0557[c]
Juvenile chinook salmon	0.012	0.000501	85	1	0.00000105
Sculpin	0.019	0.000794	79	5	0.00000834
Peamouth	0.104	0.00435	79	5	0.0000456
Smallmouth bass	0.372[d]	0.0155[d]	74	1	0.0000404
Northern pikeminnow	0.558	0.0233	74	1	0.0000606

[a] Sediment ingestion and sediment ingestion rates for fish receptors have been updated from Ecological PRE to reflect more realistic portions of incidental sediment ingestion based on discussions with fisheries experts (see Section 4.3.12).

[b] Dietary dose assessment for pre-breeding sturgeon has been added. Body weight was estimated using an allometric model, incorporating the range of sturgeon body lengths targeted for Round 3 tissue collection.

[c] Two sediment ingestion rates were evaluated for pre-breeding white sturgeon (SI = 8% and SI = 56%).

[d] Body weight and food ingestion rate have been updated from the Ecological PRE to reflect smallmouth bass collected only from within the Study Area (RM 2 to RM 11).

BW – body weight

dw – dry weight

FIR – food ingestion rate

SI – sediment ingestion

SIR – sediment ingestion rate (SIR = FIR * SI)

ww – wet weight

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Table 4-23. Prey Portions Used for Fish Dietary Risk Calculations

Receptor Species	Prey Item	Percent of Prey Tissue in Diet	Notes
Largescale sucker	worm (laboratory bioaccumulation tissue)[a]	50%	1
	clam (field-collected)	50%	
White sturgeon (pre-breeding)	worm (laboratory bioaccumulation tissue)[a]	88%	
	clam (field-collected)	12%	
Juvenile chinook salmon	multiplate invertebrate tissue[a,]	100% (Scenario 1)[b]	1
	stomach contents (Round 2)	100% (Scenario 2)[b]	
Sculpin	worm (laboratory bioaccumulation tissue)[a]	40%	1
	clam (field-collected)	40%	
	sculpin[c]	20%	
Peamouth	multiplate invertebrate tissue[a]	40%	1
	worm (laboratory bioaccumulation tissue)[a]	25%	
	clam (field-collected)	25%	
	sculpin[c]	10%	
Smallmouth bass	sculpin[c]	90%	2
	crayfish[c]	5%	
	worm (laboratory bioaccumulation tissue)[a]	5%	
Northern pikeminnow	crayfish[c]	30%	1
	sculpin[c]	25%	
	worm (laboratory bioaccumulation tissue)[a]	25%	
	largescale sucker[c]	5%	
	carp[c]	5%	
	peamouth[c, d]	5%	
	northern pikeminnow[c]	5%	

[a] Laboratory worm concentrations of PAHs, mercury, and TBT were used as a surrogate for multiplate invertebrate concentrations.

[b] Two dietary scenarios were evaluated for juvenile chinook salmon. In scenario 1, dietary concentrations were represented using multiplate prey tissue concentrations and in scenario 2, dietary concentrations were represented using juvenile chinook stomach content tissue concentrations.

[c] Because TBT was only analyzed in clams and laboratory worms, laboratory worm data were conservatively used as a surrogate for all other prey because TBT concentrations were higher in worm samples than clam samples.

[d] Sculpin concentrations of PAHs were used as surrogates for peamouth concentrations.

Notes:

1 Prey portions were updated from the Ecological PRE based on Round 2 invertebrate tissue.

2 Two diet scenarios were presented in Ecological PRE; the diet scenario representative of the Round 1 smallmouth bass size class was selected.

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Table 4-24. EPCs Based on UCL Concentrations in Prey Tissue for Dietary Round 2 COPCs

Prey Item	EPC							
	Metals (mg/kg ww)			Organics (µg/kg ww)				
	Cadmium	Copper	Mercury	TBT	Benzo(a) pyrene	Total PAHs	Total PCBs	Total DDTs
Laboratory worm	0.083	3.8	0.0060	540	240	16,000	2,390	613
Clam (field-collected)	0.11	10	0.010	220	610	891	576	130
Multiplate tissue	0.0366[a]	6.0[a]	0.0060[b]	540[b]	610[b]	16,000[b]	393	62.0
Crayfish	0.020	15	0.030	540[c]	17	410	170	27
Largescale sucker	0.026	1.0	0.081	540[c]	16	770	1,540	624
Carp	0.086	1.29	0.045	540[c]	16	930	6,470	310
Sculpin	0.011	1.3	0.047	540[c]	16	380	1,300	1,400
Peamouth	0.049	1.7	0.051	540[c]	16[d]	380[d]	279	214
Northern pikeminnow	0.011	0.76	0.40	540[c]	16[e]	680[e]	1,350	593
Chinook salmon stomach contents	NA	NA	NA	NA	NA	3,000	148	242

[a] The maximum concentration was used because insufficient number of unique values were available to calculate UCL concentration.
[b] Laboratory worm data for mercury, TBT, and total PAHs were used as surrogates because these chemicals were not analyzed in multiplate invertebrate tissue.
[c] Laboratory worm data were used as a surrogate for all prey species where TBT was not analyzed.
[d] Sculpin data for PAHs were used as a surrogate because this chemical was not analyzed for in peamouth.
[c] Smallmouth bass data for PAHs were used as a surrogate because this chemical was not analyzed for in northern pikeminnow.

COPC – chemical of potential concern
EPC – exposure point concentration
NA – not available; COPCs were not analyzed
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TBT – tributyltin
ww – wet weight

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Table 4-25. EPCs Based on UCL Concentrations in Surface Sediment for Dietary Round 2 COPCs

Round 2 COPC	Unit	EPC
Cadmium	mg/kg dw	0.69
Copper	mg/kg dw	67
Mercury	mg/kg dw	0.12
Selenium	mg/kg dw	2.3
Tributyltin	µg/kg dw	460
Benzo(a)pyrene	µg/kg dw	4,600
Total PAHs	µg/kg dw	78,000
Total DDTs	µg/kg dw	340
Total PCBs	µg/kg dw	400

COPC – chemical of potential concern
dw – dry weight
EPC – exposure point concentration
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
UCL – upper confidence limit

Table 4-26. Estimated Round 2 COPC IR_{dose} for Fish Receptors

	IR_{dose}							
	Metals (mg/kg bw/day)			Organics (µg/kg bw/day)				
Fish Receptor	Cadmium	Copper	Mercury	TBT	Benzo(a)-pyrene	Total PAHs	Total DDTs	Total PCBs
Largescale sucker	0.0044	0.32	NE	16	NE	NE	16	62
Pre-breeding sturgeon – Scenario 1	0.0039	0.22	0.00033	21	25.9	631	23	91
Pre-breeding sturgeon – Scenario 2	0.0060	0.42	0.00069	23	39.7	865	24	92
Sculpin	0.0036	0.27	NE	17	NE	NE	24	61
Peamouth	0.0030	0.28	NE	19	NE	NE	15	43
Juvenile chinook salmon – Scenario 1	0.0016	0.26	NE	23	NE	NE	NE	16
Juvenile chinook salmon – Scenario 2	NA	NA	NA	NA	NE	NE	NE	6.2
Smallmouth bass	0.00070	0.10	NE	23	NE	NE	54	54
Northern pikeminnow	0.0017	0.26	0.0021	23	NE	NE	25	61

bw – body weight
COPC – chemical of potential concern
IR – ingestion rate
NA – not analyzed
NE – not evaluated; not identified as a Round 2 COPC/receptor pair
PAH – polycyclic aromatic hydrocarbon
TBT – tributyltin

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Table 4-27. Cadmium Dietary Toxicity Studies for Fish

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Effect Endpoints	Source
Cadmium nitrate	rockfish (juvenile)	**0.0020**[a]	**0.010**	fed prepared diet for 60 days	reduced body weight and length, growth rate, and condition factor	Kim et al. (2004); Kang et al. (2005)
Cadmium chloride	guppy (adult)	1.2		fed midge larvae per day for 2 months	no effect on number of live fry, fry survival, or premature release of embryos	Hatakeyama and Yasuno (1987)
Cadmium chloride	guppy (60 days old)	1.6		fed midge larvae per day for 30 days	no effect on body weight	Hatakeyama and Yasuno (1987)
Cadmium nitrate	rockfish	2.5		fed prepared diet per day for 60 days	no effect on survival	Kim et al. (2004)
Cadmium chloride	guppy (30 days old)	3.2	4.6	fed live midge larvae per day for 7 months	reduced cumulative number of fry produced	Hatakeyama and Yasuno (1987)
Cadmium chloride	rainbow trout	4.1		fed of live brine shrimp per day for 60 days[b]	no effect on survival	Mount et al. (1994)
Cadmium nitrate	rainbow trout	5.9		fed prepared diet per day for 15 to 30 days	no effect on specific growth rate, survival	Baldisserotto et al. (2005)
Cadmium chloride	guppy	1.0		fed M*oina macrocoipa* per day for 10 to 30 days	no effect on growth[c]	Hatakeyama and Yasuno (1982)
Cadmium nitrate	rainbow trout	9.4		fed prepared diet per day for 28 days	no effect on growth rate or survival	Franklin et al. (2005)
Cadmium chloride	guppy (30 days old)		9.4	fed live midge larvae per day for 7 months	reduced female growth and survival	Hatakeyama and Yasuno (1987)
Cadmium sulfate	juvenile rainbow trout		68	fed prepared diet per day for 28 days	reduced survival	Handy (1992)

[a] NOAEL extrapolated from LOAEL using a UF of 5.
[b] Fish were exposed to cadmium in both water and the diet.
[c] A significant effect on growth was observed at day 10; however, growth was recovered by day 20.
bw – body weight
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
Bold identifies the NOAEL and LOAEL selected as dietary TRVs.

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Table 4-28. Copper Dietary Toxicity Studies for Fish

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effects	Source
Copper sulfate	channel catfish (fingerling)	**0.24**	**0.48**	fed prepared diet per day for 16 weeks	reduced growth	Murai et al. (1981)
Copper sulfate	rockfish (juvenile)	1.0	2.0	fed prepared diet for 60 days	reduced growth rate	Kang et al. (2005)
Copper sulphate	rainbow trout	2.2		fed prepared diet for 32 days	no effect on survival	Handy (1992)
Copper sulphate pentahydrate	rainbow trout	6.8		fed prepared diet for 16 weeks	no effect on growth	Lanno et al. (1985b)
Copper sulfate	rainbow trout	13		fed prepared diet for 42 days	no effect on growth	Miller et al. (1993)
Copper sulphate pentahydrate	Atlantic salmon (parr)	17		fed prepared diet for 4 weeks	no effect on body weight, length, or condition factor	Berntssen et al. (1999b)
Copper sulphate pentahydrate	rainbow trout	17		fed prepared diet for 24 weeks	no effect on growth or survival	Lanno et al. (1985b)
Copper sulphate pentahydrate	rainbow trout	18		fed prepared diet for 8 weeks	no effect on survival	Lanno et al. (1985b)
Copper sulphate pentahydrate	rainbow trout		18	fed prepared diet for 8 weeks	reduced growth	Lanno et al. (1985b)
Copper sulphate pentahydrate	rainbow trout		20	fed prepared diet for 16 weeks	reduced growth[a]	Lanno et al. (1985a)
Cu sulfate pentahydrate	Atlantic salmon (fry)	14	20	fed prepared diet for 3 months	reduced growth	Lundebye et al. (1999)
Copper sulphate	Atlantic salmon	20	28	fed prepared diet for 3 months	reduced growth	Berntssen et al. (1999a)
Copper sulphate pentahydrate	rainbow trout	42		fed prepared diet for 28 days	no effect on growth or survival	Kamunde et al. (2001)
Copper chloride	rainbow trout (fry)	26[b]	50[b]	fed brine shrimp for 60 days	reduced survival	Mount et al. (1994)
Copper chloride	rainbow trout (fry)	60[b]		fed brine shrimp for 60 days	no effect on body weight and length	Mount et al. (1994)
Copper sulphate pentahydrate	grey mullet		60	fed prepared diet for 67 days	reduced growth[c]	Baker et al. (1998)
Copper sulphate	rainbow trout	69		fed prepared diet for 28 days	no effect on survival	Handy (1993)

[a] A significant reduction in growth was observed at 16 weeks; however, growth was recovered by 24 weeks.
[b] Fish were exposed to copper in both water and diet.
[c] Observed reduction of growth was associated with a reduction in feeding.
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
Bold identifies the NOAEL and LOAEL selected as dietary TRVs.

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Table 4-29. Mercury Dietary Toxicity Studies for Fish

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Effect Endpoint	Source
Methyl-mercury	golden shiner		0.0091	fed prepared diet for 90 days	altered shoaling behavior after predator exposure	Webber and Haines (2003)
Methyl-mercury	golden shiner	0.019		fed prepared diet for 90 days	no effect on growth or survival	Webber and Haines (2003)
Methyl-mercury	walleye	0.0036[a]	0.026[a]	fed catfish fillets for 6 months	reduced male growth	Friedmann et al. (1996)
Methyl-mercuric chloride	mummichog	**0.013**	**0.048**	fed prepared diet for at least 6 weeks	reduced male survival	Matta et al. (2001)
Methyl-mercury	fathead minnow		0.053	fed prepared diet for multiple generations	reduced spawning success	Hammerschmidt et al. (2002)
Methyl-mercury	walleye		0.053	fed catfish fillets for 6 months	reduced male gonadosomatic index	Friedmann et al. (1996)
Methyl-mercury	rainbow trout		0.46	fed prepared diet for 12 weeks	reduced body weight in hatchling fish	Rodgers and Beamish (1982)
Methyl-mercuric chloride	mummichog	0.14	1.4	fed prepared diet for at least 6 weeks	reduced fertilization success	Matta et al. (2001)
Methyl-mercuric chloride	mummichog	1.4		fed prepared diet for at least 6 weeks	no effect on fertilization success, fecundity, offspring hatchability, survival, or body weight, or on adult female survival	Matta et al. (2001)
Methyl-mercury	rainbow trout	1.5		fed prepared diet for 12 weeks	no effect on survival	Rodgers and Beamish (1982)
Methyl-mercury	rainbow trout		1.5	fed prepared diet for 12 weeks	lethargic behavior and darker appearance of fish	Rodgers and Beamish (1982)

[a] Elevated mortality reported in control group; no replication; high uncertainty associated with estimated dietary dose.

bw – body weight

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

Bold identifies the NOAEL and LOAEL selected as dietary TRVs.

Table 4-30. Tributyltin Dietary Toxicity Studies for Fish

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Effect Endpoints	Source
Tributyltin oxide	Japanese flounder (larvae)	**0.0021**	**0.020**	fed prepared diet for approximately 65 days	reduced body weight, increased sex reversal	Shimasaki et al. (2003)

bw – body weight
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
Bold identifies the NOAEL and LOAEL selected as dietary TRVs.

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Table 4-31. PAH Mixture Dietary Toxicity Studies for Fish

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Effect Endpoint	Source
Benzo(a)pyrene	juvenile English sole	**0.66**	**1.4**	fed contaminated worms for 10 to 12 days	reduced daily growth rate	Rice et al. (2000)
Benzo(a)pyrene	juvenile rainbow trout		3.7	fed prepared diet for 53 days	reduced body weight; no effect on survival	Hendricks et al. (1985)
Benzo(a)pyrene	juvenile rainbow trout	1.9	19	fed prepared diet for 18 months[a]	reduced body weight	Hart and Heddle (1991)
Benzo(a)pyrene	grouper	13		force-fed PAH contaminated pellet for 4 weeks	no effect on survival or growth (body weight and total length)	Wu et al. (2003)
PAH mixture	English sole	0.012	0.12	fed contaminated worms for 10-12 days[b]	reduced daily growth rate	Rice et al. (2000)
PAH mixture	juvenile chinook salmon	5.0		fed prepared diet for 7 weeks	no effect on body weight, length, or condition factor	Palm et al. (2003)
PAH mixture	juvenile chinook salmon	5.0		fed prepared diet for 4 weeks[c]	no impact to disease resistance	Palm et al. (2003)
PAH mixture	juvenile chinook salmon	**6.1**	**18.0**	fed prepared diet for 53 days	reduced body weight (as dry body weight)[d]	Meador et al. (2006)

[a] Feeding rate was estimated as 1.9% body weight per day based on the average feeding rate reported in other dietary toxicological studies were rainbow trout were fed a synthetically prepared diet (i.e. not live prey).

[b] Worms were exposed to sediments collected from a Superfund site contaminated with PAHs and potentially other chemicals that were not characterized in this study.

[c] Fish were exposed to *Listonella anguillarum* following PAH exposure. No difference was observed between PAH-exposed fish and controls in either fish that were vaccinated against the bacterium or those that were not vaccinated.

[d] Effects on growth were not significant at the LOAEL when evaluated on a wet-weight basis.

bw – body weight

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

Bold identifies the NOAEL and LOAEL selected as dietary TRVs.

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Table 4-32. DDTs Dietary Toxicity Studies for Fish

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effects	Source
p,p′-DDT	Atlantic menhaden	0.0023		fed prepared diet for 48 days	no effect on growth	Warlen et al. (1977)
p,p′-DDE	brook trout (fry)		0.0044[a]	fed prepared diet for 176 days	reduced fry survival	Berlin et al. (1981)
Total DDT	goldfish	0.019		fed prepared diet for 58 days	no effect on survival	Rhead and Perkins (1984)
Total DDT	brook trout (yearling)	**0.028**[b]	**0.14**	fed prepared diet for 156 days	reproduction	Macek (1968a)
Total DDT	rainbow trout (juvenile)	0.036		fed prepared diet for 26 weeks	no effect on final body weight, survival, or behavior	Hilton et al. (1983)
Total DDT	brook trout (juvenile)	0.045		fed prepared diet for 120 days	no effect on survival	Macek and Korn (1970)
Total DDT	rainbow trout	0.14		fed prepared diet for 168 days	no effect on survival or growth	Macek et al. (1970)
Total DDT	brook trout (sub-yearling)		0.29	fed prepared diet for 26 weeks	increased growth; reduced survival during stress (starvation)	Macek (1968a)
Technical grade DDT	brook trout (fry)	0.3		fed prepared diet for 168 days	no effect on survival or growth	Macek (1968a)
Total DDT	pinfish		0.81	fed oysters for 15 days	reduced survival	Butler (1969)
Total DDT	juvenile chinook salmon	0.48	1.16	fed prepared diet for up to 95 days	reduced survival	Buhler et al. (1969)
DDT mixture	pinfish		1.4	fed oysters for 15 days	reduced survival	Butler (1969)
Total DDT	juvenile coho salmon	0.63	2.50	fed prepared diet for up to 95 days	reduced survival	Buhler et al. (1969)
Total DDT	fathead minnow		2.7	fed dried clams for 266 days	reduced survival	Jarvinen et al. (1976; 1977)
Total DDT	juvenile chinook salmon	0.75	3.00	fed prepared diet for up to 95 days	reduced survival	Buhler et al. (1969)

[a] Study was conducted with field-collected eggs from Lake Michigan with starting egg residues of 7.6 µg/g PCBs and 4.7 µg/g DDE.
[b] NOAEL extrapolated from LOAEL using a UF of 5.
bw – body weight
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
Bold identifies the NOAEL and LOAEL selected as dietary TRVs.

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Table 4-33. PCBs Dietary Toxicity Studies for Fish

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Effect Endpoint	Source
Aroclor 1254	brook trout (fry) [a]		0.018	fed prepared diet for 176 days	reduced fry survival	Berlin et al. (1981)
Aroclor 1260	barbel (adult)	**0.010**[b]	**0.050**	fed prepared diet for 50 days	reduced fecundity	Hugla and Thome (1999)
Aroclor 1260	barbell (adult)		0.25	fed prepared diet for 50 days	failure to spawn	Hugla and Thome (1999)
Aroclor 1254	juvenile chinook salmon	0.34		fed prepared diet for 4 weeks	no effect on growth	Powell et al. (2003)
Aroclor 1254	Atlantic croaker (adult)		0.40	fed ground shrimp for 2 weeks	impaired larval response to startle stimulus	McCarthy et al. (2003)
Aroclor 1260	channel catfish (fingerling)	0.48		gavage for 193 days	no effect on growth or survival	Mayer et al. (1977)
Aroclor 1242	channel catfish (fingerling)		0.6	fed prepared diet for 20 weeks	reduced body weight gain	Hansen et al. (1976)
Aroclor 1254	rainbow trout (juvenile)	0.75		fed prepared diet for ~260 days	no effect on growth or survival	Lieb et al. (1974)
Aroclor 1268	mummichog (adult)	0.800		fed prepared diet for at least 6 weeks	no effect on fertilization, hatching success, larval survival, or larval weight	Matta et al. (2001)
Aroclor 1254	rainbow trout (adult)		3.80	fed prepared diet for 60 days	reduced offspring fry growth	Hendricks et al. (1981)
Aroclor 1254	coho salmon (fingerling)	1.5	14.5	gavage for ~260 days	reduced survival[c]	Mayer et al. (1977)

[a] Study was conducted with field-collected eggs from Lake Michigan with starting egg residues of 7.6 µg/g PCBs and 4.7 µg/g DDE.

[b] NOAEL extrapolated from LOAEL using a UF of 5.

[c] All fish died within 265 days of dose in LOAEL group. No control group was presented in study.

bw – body weight

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

Bold identifies the NOAEL and LOAEL selected as dietary TRVs.

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Table 4-34. Summary of Surface Water EPCs for Fish

Round 2 COPC	Unit	EPC	
		Peristaltic Pump Samples	XAD Samples
Zinc (dissolved)	mg/L	0.0049	NA
Benzo(a)anthracene	µg/L	0.011	0.0017
Benzo(a)pyrene	µg/L	0.013	0.00167
4-Chloro-3-methylphenol	µg/L	0.1	NA
Total PCBs	µg/L	0.0051	0.00325
2,4′-DDD	µg/L	0.00025	0.0014
2,4′-DDT	µg/L	0.0021	0.00031
4,4′-DDD	µg/L	0.00028	0.0027
4,4′-DDT	µg/L	0.00031	0.00212
Total DDTs	µg/L	0.0024	0.00672

COPC – chemical of potential concern
NA – not analyzed
PCB – polychlorinated biphenyl
UCL – upper confidence limit

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Table 4-35. Selected Chronic and Acute Eco SLs for Surface Water Round 2 COPCs

Chemical Form	Unit	Chronic Eco SL	Acute Eco SL	Source	Comments
Zinc (dissolved)	mg/L	0.0365	0.0362	AWQC	Criteria are hardness-dependent. These values correspond to a hardness of 25 mg/L calcium carbonate (as estimated for LWR) and were adjusted using EPA-provided equations. Criteria are for dissolved fraction.
Benzo(a)anthracene	µg/L	0.027	0.49	Tier II	
Benzo(a)pyrene	µg/L	0.014	0.24	Tier II	
4-Chloro-3-methylphenol	µg/L	0.6	30	ODEQ	Acute Eco SL was divided by a UF of 50 to calculate the chronic Eco SL.
Total PCBs	µg/L	0.014	2	ODEQ (acute); ODEQ and Tier II (chronic)	Total PCBs criterion applies to total PCBs, as either sum of all homologs, Aroclors, or congeners.
2,4′-DDD	µg/L	0.001	1.1	AWQC	AWQC for 4,4′-DDT applies to 2,4′-DDD.
4,4′-DDD	µg/L	0.001	1.1	AWQC	AWQC for 4,4′-DDT applies to 4,4′-DDD.
2,4′-DDT	µg/L	0.001	1.1	AWQC	AWQC for 4,4′-DDT applies to 2,4′-DDT.
4,4′-DDT	µg/L	0.001	1.1	AWQC	
Total DDTs	µg/L	0.001	1.1	AWQC	AWQC for 4,4′-DDT applies to total DDTs.

AWQC – ambient water quality criteria
COPC – chemical of potential concern
Eco SL – Ecological Screening Level
EPA – US Environmental Protection Agency
LWR – Lower Willamette River
ODEQ – Oregon Department of Environmental Quality
PCB – polychlorinated biphenyl

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Table 4-36. Potential TZW iCOCs for Fish

Potential iCOCs	
PAHs	
2-Methylnaphthalene	Chrysene
Acenaphthene	Dibenzo(a,h)anthracene
Anthracene	Fluoranthene
Benzo(a)anthracene	Fluorene
Benzo(a)pyrene	Indeno(1,2,3-cd)pyrene
Benzo(b)fluoranthene	Naphthalene
Benzo(g,h,i)perylene	Phenanthrene
Benzo(k)fluoranthene	Pyrene
Pesticides	
2,4′-DDD	4,4′-DDE
2,4′-DDT	4,4′-DDT
4,4′-DDD	Total DDTs
Cyanide	
Cyanide	

iCOC – initial chemical of concern
PAH – polycyclic aromatic hydrocarbon
TZW – transition zone water

Table 4-37. Largescale Sucker HQs Based on the Tissue-Residue Line of Evidence

Sample ID	Chromium			BEHP			Total PCBs			4,4′-DDD			Total DDTs		
	EPC (µg/kg)	NOAEL HQ	LOAEL HQ	EPC (µg/kg)	NOAEL HQ	LOAEL HQ	EPC (µg/kg)	NOAEL HQ	LOAEL HQ	EPC (µg/kg)	NOAEL HQ	LOAEL HQ	EPC (µg/kg)	NOAEL HQ	LOAEL HQ
UCL (all locations)	1.9	0.35	0.22	5,500	**14**	0.52	1,540	0.81	0.38	110	NC	NC	624	0.35	0.35
03R014	0.78	0.14	0.090	78 UT	0.20	0.0074	2,060 J	**1.1**	0.51	41 T	NC	NC	228 J	0.13	0.13
03R014 (FR)	0.38	0.069	0.044	99 UT	0.25	0.0093	358	0.19	0.089	31 J	NC	NC	176 NJ	0.098	0.098
05R006	0.60	0.11	0.069	97 UT	0.25	0.0092	96.9	0.051	0.024	20	NC	NC	152 NJ	0.084	0.084
07R009	0.41	0.075	0.047	78 UT	0.20	0.0074	1,650	0.87	0.41	57 T	NC	NC	196 J	0.11	0.11
08R010	2.77 T	0.50	0.32	3,000 JT	**7.7**	0.28	525	0.28	0.13	150 T	NC	NC	673	0.37	0.37
09R006	1.76	0.32	0.20	800 T	**2.1**	0.075	738	0.39	0.18	27.5 JT	NC	NC	155 J	0.086	0.086

BEHP – bis(2-ethylhexyl) phthalate
EPC – exposure point concentration
FR – field replicate
HQ – hazard quotient
ID – identification
J – estimated concentration
LOAEL – lowest-observed-adverse-effect level
N – presumptive evidence of a compound
Bold identifies HQs greater than 1.0.
NC – not calculated; no LOAEL or NOAEL TRV recommended by LWG
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
T – value calculated or selected from multiple results
TRV – toxicity reference value
U – not detected at given concentration
UCL – upper confidence limit

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Table 4-38. Sculpin HQs for DDT Based on the Tissue-Residue Line of Evidence

Sample ID	4,4′-DDD			4,4′-DDT			Total DDTs		
	EPC (µg/kg)	NOAEL HQ	LOAEL HQ	EPC (µg/kg)	NOAEL HQ	LOAEL HQ	EPC (µg/kg)	NOAEL HQ	LOAEL HQ
UCL (all samples)	150	NC	NC	750	NC	NC	1,400	0.78	0.78
02R001	14 JT	NC	NC	6.3 UT	NC	NC	56.4 J	0.031	0.031
02R015	9.2 JT	NC	NC	6.3 UT	NC	NC	51.6 J	0.029	0.029
03R001	5.2 NJ	NC	NC	16 J	NC	NC	40.6 NJ	0.023	0.023
03R002	6.7 N	NC	NC	19	NC	NC	45.9 NJ	0.026	0.026
03R002 (FR)	7.2 N	NC	NC	37	NC	NC	72.2 NJ	0.040	0.040
03R004	5.7 N	NC	NC	54	NC	NC	102 N	0.057	0.057
03R004 (FR)	8.3 N	NC	NC	50	NC	NC	110 NJ	0.061	0.061
03R005	7.7 N	NC	NC	75	NC	NC	144 NJ	0.080	0.080
03R032	7.3 N	NC	NC	23 J	NC	NC	57.1 NJ	0.032	0.032
03R034	6.1 UT	NC	NC	12 JT	NC	NC	46.5 J	0.026	0.026
04R002	5.6 J	NC	NC	26 J	NC	NC	63.2 NJ	0.035	0.035
04R003	5.3 N	NC	NC	21 J	NC	NC	51.3 NJ	0.029	0.029
04R004	12 J	NC	NC	26 J	NC	NC	77.1 NJ	0.043	0.043
05R001	6.1 UT	NC	NC	6.3 UT	NC	NC	31.7 J	0.018	0.018
05R020	11 J	NC	NC	22 J	NC	NC	50.3 J	0.028	0.028
06R001	16 J	NC	NC	27 J	NC	NC	82.8 NJ	0.046	0.046
06R002	6.1 UT	NC	NC	6.3 UT	NC	NC	32.7 J	0.018	0.018
06R002 (FR)	12 JT	NC	NC	11 JT	NC	NC	60.5 J	0.034	0.034
06R004	140 JT	NC	NC	135 JT	NC	NC	570 J	0.32	0.32
07R003	84 T	NC	NC	250 T	NC	NC	582 J	0.32	0.32
07R006	305 T	NC	NC	1,700 T	NC	NC	3,060	**1.7**	**1.7**
08R001	6.1 UT	NC	NC	8.9 JT	NC	NC	52.5 J	0.029	0.029
08R002	4.6 NJ	NC	NC	21 J	NC	NC	45.7 NJ	0.025	0.025
08R003	6.1 UT	NC	NC	6.3 UT	NC	NC	31.7 J	0.018	0.018

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Table 4-38. Sculpin HQs for DDT Based on the Tissue-Residue Line of Evidence

Sample ID	4,4′-DDD			4,4′-DDT			Total DDTs		
	EPC (µg/kg)	NOAEL HQ	LOAEL HQ	EPC (µg/kg)	NOAEL HQ	LOAEL HQ	EPC (µg/kg)	NOAEL HQ	LOAEL HQ
09R001	4 UT	NC	NC	19 JT	NC	NC	52.5 J	0.029	0.029
09R002	9.6 U	NC	NC	81	NC	NC	182 NJ	0.10	0.10

EPC – exposure point concentration
FR – field replicate
HQ – hazard quotient
ID – identification
J – estimated concentration
LOAEL – lowest-observed-adverse-effect level
N – presumptive evidence of a compound
NC – not calculated; no LOAEL or NOAEL TRV recommended by LWG
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
T – value calculated or selected from multiple results
TRV – toxicity reference value
U – not detected at given concentration
UCL – upper confidence limit
Bold identifies HQs greater than 1.0.

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Table 4-39. Sculpin HQs for Total PCBs, beta-HCH, and BEHP Based on the Tissue-Residue Line of Evidence

Sample ID	Total PCBs			beta-HCH			BEHP		
	EPC (µg/kg)	NOAEL HQ	LOAEL HQ	EPC (µg/kg)	NOAEL HQ	LOAEL HQ	EPC (µg/kg)	NOAEL HQ	LOAEL HQ
UCL (all samples)	1,300	0.68	0.32	4.8	0.0030	0.000061	13,000	**33**	**1.2**
02R001	2,370 J	**1.2**	0.59	4.0 UT	0.0025	0.000051	92 UT	0.24	0.0087
02R015	3,400	**1.8**	0.85	8.5 UT	0.0054	0.000108	400 UT	1.0	0.038
03R001	146 J	0.077	0.036	4.3 NJ	0.0027	0.000054	150 UT	0.38	0.014
03R002	123 J	0.065	0.031	4.1 NJ	0.0026	0.000052	180 UT	0.46	0.017
03R002 (FR)	172 J	0.091	0.043	6.2 NJ	0.0039	0.000078	96 UT	0.25	0.0091
03R004	334	0.18	0.083	4.5 NJ	0.0028	0.000057	810 UT	**2.1**	0.076
03R004 (FR)	315	0.17	0.078	2.9 NJ	0.0018	0.000037	140 UT	0.36	0.013
03R005	370	0.19	0.092	1.0 UJ	0.00063	0.000013	160 UT	0.41	0.015
03R032	176	0.093	0.044	4.1 NJ	0.0026	0.000052	86 UT	0.22	0.0081
03R034	538	0.28	0.13	4.8 UT	0.0030	0.000061	150 UT	0.38	0.014
04R002	166 J	0.087	0.041	3.8 NJ	0.0024	0.000048	240 UT	0.62	0.023
04R003	198 J	0.10	0.049	2.5 NJ	0.0016	0.000032	260 UT	0.67	0.025
04R004	125 J	0.066	0.031	3.4 NJ	0.0022	0.000043	9,400 JT	**24**	0.89
05R001	198 J	0.10	0.049	8.5 UT	0.0054	0.000108	96 UT	0.25	0.0091
05R020	134 J	0.071	0.033	3.9 NJ	0.0025	0.000049	170 UT	0.44	0.016
06R001	100 J	0.053	0.025	5.3 NJ	0.0034	0.000067	82 UT	0.21	0.0077
06R002	691 J	0.36	0.17	8.5 UT	0.0054	0.000108	99 T	0.25	0.0093
06R002 (FR)	2,450	**1.3**	0.61	8.5 UT	0.005408	0.000108	260 UT	0.67	0.025
06R004	588 J	0.31	0.15	8.5 UT	0.0054	0.000108	200 UT	0.51	0.019
07R003	236 J	0.12	0.059	8.5 UT	0.0054	0.000108	100 UT	0.26	0.0094
07R006	440 J	0.23	0.11	1.0 UJT	0.00063	0.000013	220 UT	0.56	0.021
08R001	189	0.099	0.047	8.5 UT	0.0054	0.000108	160 UT	0.41	0.015
08R002	159	0.084	0.04	2.0 NJ	0.0013	0.000025	100 UT	0.26	0.0094

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Table 4-39. Sculpin HQs for Total PCBs, beta-HCH, and BEHP Based on the Tissue-Residue Line of Evidence

Sample ID	Total PCBs			beta-HCH			BEHP		
	EPC (µg/kg)	NOAEL HQ	LOAEL HQ	EPC (µg/kg)	NOAEL HQ	LOAEL HQ	EPC (µg/kg)	NOAEL HQ	LOAEL HQ
08R003	583 J	0.31	0.15	4.0 UT	0.0025	0.000051	28,000 JT	**72**	**2.6**
09R001	651 J	0.34	0.16	8.5 UT	0.0054	0.000108	220 UT	0.56	0.021
09R002	823	0.43	0.20	9.6 U	0.0061	0.000122	120 UT	0.31	0.011

BEHP – bis(2-ethylhexyl) phthalate
EPC – exposure point concentration
FR – field replicate
HCH –hexachlorocyclohexane
HQ – hazard quotient
ID – identification
J – estimated concentration
LOAEL – lowest-observed-adverse-effect level
N – presumptive evidence of a compound
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
T – value calculated or selected from multiple results
TRV – toxicity reference value
U – not detected at given concentration
UCL – upper confidence limit

Bold identifies HQs greater than 1.0.

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Table 4-40. Sculpin HQs Based on the Predicted Tissue-Residue Line of Evidence

Round 2 COPC	Unit (ww)	EPC[a]	NOAEL TRV	LOAEL TRV	NOAEL HQ	LOAEL HQ
Selenium	mg/kg	1.73	NA	1.58	NA	**1.1**
BEHP	mg/kg lipid	1,690	9.35	254	**180**	**6.7**

[a] EPC based on predicted tissue concentration calculated from a site-specific BSAF and the site-wide UCL sediment concentration.

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
EPC – exposure point concentration
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
TRV – toxicity reference value
ww – wet weight
Bold identifies HQs greater than 1.0.

Table 4-41. Peamouth HQs Based on the Tissue-Residue Line of Evidence

Sample ID	Lead		
	EPC (mg/kg)	NOAEL HQ	LOAEL HQ
UCL (all samples)	37	**15**	**9.2**
03R014	10.6	**4.2**	**2.6**
05R006	0.111	0.044	0.028
08R010	0.047	0.019	0.012
09R006	0.031	0.012	0.0077

EPC – exposure point concentration
HQ – hazard quotient
ID – identification
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
Bold identifies HQs greater than 1.0.

Table 4-42. Juvenile Chinook Salmon HQs Based on the Tissue-Residue Line of Evidence

Sample ID	Zinc			4,4′-DDT		
	EPC (mg/kg)	NOAEL HQ	LOAEL HQ	EPC (mg/kg)	NOAEL HQ	LOAEL HQ
UCL (all samples)	30.7	0.90	0.68	150	NC	NC
T01-REP1	27.9 T	0.82	0.70	1.0 J	NC	NC
T01-REP2	25.935	0.76	0.65	1.9	NC	NC
T01-REP3	27.676	0.81	0.69	1.6	NC	NC
T02-REP1	30.81	0.91	0.77	110	NC	NC
T02-REP2	29.008	0.85	0.73	120	NC	NC
T02-REP3	29.1	0.86	0.73	130	NC	NC
T03-REP1	24.038	0.71	0.60	1.2 J	NC	NC
T03-REP2	31.52	0.93	0.79	1.5 J	NC	NC
T03-REP3	28.665	0.84	0.72	1.4 J	NC	NC
02R102	31.1	0.91	0.78	4.6 N	NC	NC
02R112	33.3	0.98	0.83	6 JT	NC	NC
02R113	33.3	0.98	0.83	5.2 JT	NC	NC
03R118	29.6	0.87	0.74	4.2 J	NC	NC
03R125	29.6	0.87	0.74	6.3 J	NC	NC
04R126	31.5	0.93	0.79	8.1 J	NC	NC

EPC – exposure point concentration
FR – field replicate
HQ – hazard quotient
ID – identification
J – estimated concentration
LOAEL – lowest-observed-adverse-effect level
Bold identifies HQs greater than 1.0.
N – presumptive evidence of a compound
NC – not calculated; no LOAEL or NOAEL TRV recommended
NOAEL – no-observed-adverse-effect level
T – value calculated or selected from multiple results
TRV – toxicity reference value
UCL – upper confidence limit

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Table 4-43. Smallmouth Bass HQs Based on the Tissue-Residue Line of Evidence

Sample ID	Total PCBs			4,4′-DDD			Total DDTs			BEHP		
	EPC (µg/kg)	NOAEL HQ	LOAEL HQ	EPC (µg/kg)	NOAEL HQ	LOAEL HQ	EPC (µg/kg)	NOAEL HQ	LOAEL HQ	EPC (µg/kg)	NOAEL HQ	LOAEL HQ
UCL (all samples)	1,960	1.0	0.49	51	NC	NC	262	0.15	0.15	73,000	**190**	**6.9**
03R014	788	0.41	0.20	30.5 JT	NC	NC	199 J	0.11	0.11	1,300 UT	**3.3**	0.12
04R023	1,320 J	0.69	0.33	47 T	NC	NC	309 J	0.17	0.17	32,000 JT	**82**	**3.0**
04R023 (FR)	478	0.25	0.12	33.5 JT	NC	NC	185 J	0.10	0.10	690 UT	**1.8**	0.065
04R023 (FR)	600 J	0.32	0.15	31 JT	NC	NC	140 J	0.078	0.078	87,000 JT	**220**	**8.2**
05R006	400 J	0.21	0.10	35 JT	NC	NC	185 J	0.10	0.10	92 UT	0.24	0.0087
06R024	278 J	0.15	0.069	19 JT	NC	NC	149 J	0.083	0.083	96 UT	0.25	0.0091
07R009	544	0.29	0.14	58 T	NC	NC	416	0.23	0.23	86 UT	0.22	0.0081
07R009 (FR)	109 J	0.057	0.027	55 T	NC	NC	365 J	0.20	0.20	630 UT	**1.6**	0.059
07R009 (FR)	818 J	0.43	0.20	110 T	NC	NC	341	0.19	0.19	98 UT	0.25	0.0092
08R010	4,950 J	**2.6**	**1.2**	11 JT	NC	NC	122 J	0.068	0.068	99 UT	0.25	0.0093
08R010 (FR)	3,610 J	**1.9**	0.90	12 JT	NC	NC	79.7 J	0.044	0.044	96 UT	0.25	0.0091
08R010 (FR)	1,100	0.58	0.27	25.5 T	NC	NC	122 J	0.068	0.068	94 UT	0.24	0.0089
08R032	918 J	0.48	0.23	25 JT	NC	NC	189 J	0.11	0.11	96 UT	0.25	0.0091
09R006	878 J	0.46	0.22	37.5 T	NC	NC	192	0.11	0.11	96 UT	0.25	0.0091

BEHP – bis(2-ethylhexyl) phthalate
EPC – exposure point concentration
FR – field replicate
HQ – hazard quotient
ID – identification
J – estimated concentration
LOAEL – lowest-observed-adverse-effect level
N – presumptive evidence of a compound
NC – not calculated; no LOAEL or NOAEL TRV recommended by LWG
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
T – value calculated or selected from multiple results
TRV – toxicity reference value
U – not detected at given concentration
UCL – upper confidence limit

Bold identifies HQs greater than 1.0.

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Table 4-44. Northern Pikeminnow HQs Based on the Tissue-Residue Line of Evidence

Sample ID	Total PCBs			Mercury			Total DDTs		
	EPC (µg/kg)	NOAEL HQ	LOAEL HQ	EPC (µg/kg)	NOAEL HQ	LOAEL HQ	EPC (µg/kg)	NOAEL HQ	LOAEL HQ
UCL (all samples)	1,350	0.71	0.34	0.40	**2.0**	0.85	593	0.33	0.33
03R014	729	0.38	0.18	0.173	0.87	0.37	286	0.16	0.16
03R014 (FR)	378	0.20	0.094	0.146 T	0.73	0.31	261	0.15	0.15
05R006	448	0.24	0.11	0.326	1.6	0.69	365	0.20	0.20
07R009	1,930	1.0	0.48	0.494	**2.5**	**1.1**	764	0.42	0.42
08R010	795	0.42	0.20	0.153	0.77	0.33	172 NJ	0.096	0.096
09R006	1,030	0.54	0.26	0.39	**2.0**	0.83	163	0.091	0.091

EPC – exposure point concentration
FR – field replicate
HQ – hazard quotient
ID – identification
J – estimated concentration
LOAEL – lowest-observed-adverse-effect level
Bold identifies HQs greater than 1.0.
N – presumptive evidence of a compound
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
T – value calculated or selected from multiple results
TRV – toxicity reference value
UCL – upper confidence limit

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Table 4-45. Largescale Sucker HQs Based on the Dietary Dose Line of Evidence

Round 2 COPC	Unit (bw/day)	IR Dose	NOAEL TRV	LOAEL TRV	NOAEL HQ	LOAEL HQ
Cadmium	mg/kg	0.00438	0.0020	0.010	**2.2**	0.44
Copper	mg/kg	0.322	0.24	0.48	**1.3**	0.67
TBT	µg/kg	16.1	2.1	20	**7.7**	0.81
Total PCBs	µg/kg	62.2	10	50	**6.2**	**1.2**
Total DDTs	µg/kg	15.7	28	140	0.56	0.11

bw – body weight
COPC – chemical of potential concern
HQ – hazard quotient
IR – ingestion rate
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TBT – tributyltin
TRV – toxicity reference value
Bold identifies HQs greater than 1.0.

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Table 4-46. Pre-breeding Sturgeon HQs Based on the Dietary Dose Line of Evidence

Round 2 COPC	Unit (bw/day)	IR Dose	NOAEL TRV	LOAEL TRV	NOAEL HQ	LOAEL HQ
Scenario 1[a]						
Cadmium	mg/kg	0.00394	0.0020	0.010	**2.0**	0.39
Copper	mg/kg	0.223	0.24	0.48	0.93	0.46
Mercury	mg/kg	0.000330	0.013	0.048	0.025	0.0069
TBT	µg/kg	21.1	2.1	20	**10**	**1.1**
Benzo(a) pyrene	µg/kg	25.9	660	1,400	0.039	0.018
Total PAHs	µg/kg	631	6,100	18,000	0.10	0.035
Total PCBs	µg/kg	90.8	10	50	**9.1**	**1.8**
Total DDTs	µg/kg	23.3	28	140	0.83	0.17
Scenario 2[b]						
Cadmium	mg/kg	0.00601	0.0020	0.010	**3.0**	0.60
Copper	mg/kg	0.424	0.24	0.48	**1.8**	0.88
Mercury	mg/kg	0.000691	0.013	0.048	0.053	0.014
TBT	µg/kg	22.5	2.1	20	**11**	**1.1**
Benzo(a) pyrene	µg/kg	39.7	660	1,400	0.060	0.028
Total PAHs	µg/kg	865	6,100	18,000	0.14	0.048
Total PCBs	µg/kg	92.0	10	50	**9.2**	**1.8**
Total DDTs	µg/kg	24.3	28	140	0.87	0.17

[a] Scenario 1 represents the dietary dose based on 8% incidental sediment ingestion.
[b] Scenario 2 represents the dietary dose based on 56% incidental sediment ingestion.
bw – body weight
COPC – chemical of potential concern
HQ – hazard quotient
IR – ingestion rate
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TBT – tributyltin
TRV – toxicity reference value
Bold identifies HQs greater than 1.0.

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Table 4-47. Sculpin HQs Based on the Dietary Dose Line of Evidence

Round 2 COPC	Unit (bw/day)	IR Dose	NOAEL TRV	LOAEL TRV	NOAEL HQ	LOAEL HQ
Cadmium	mg/kg	0.00362	0.0020	0.010	**1.8**	0.36
Copper	mg/kg	0.271	0.24	0.48	**1.1**	0.56
TBT	µg/kg	17.4	2.1	20	**8.3**	0.87
Total PCBs	µg/kg	60.6	10	50	**6.1**	**1.2**
Total DDTs	µg/kg	24.3	28	140	0.87	0.17

bw – body weight
COPC – chemical of potential concern
HQ – hazard quotient
IR – ingestion rate
LOAEL – lowest-observed-adverse-effect level
Bold identifies HQs greater than 1.0.
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TBT – tributyltin
TRV – toxicity reference value

Table 4-48. Peamouth HQs Based on the Dietary Dose Line of Evidence

Round 2 COPC	Unit (bw/day)	IR Dose	NOAEL TRV	LOAEL TRV	NOAEL HQ	LOAEL HQ
Cadmium	mg/kg	0.00298	0.0020	0.010	**1.5**	0.30
Copper	mg/kg	0.280	0.24	0.48	**1.2**	0.58
TBT	µg/kg	19.4	2.1	20	**9.3**	1.0
Total PCBs	µg/kg	43.2	10	50	**4.3**	0.86
Total DDTs	µg/kg	14.8	28	140	0.53	0.11

bw – body weight
COPC – chemical of potential concern
HQ – hazard quotient
IR – ingestion rate
LOAEL – lowest-observed-adverse-effect level
Bold identifies HQs greater than 1.0.
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TBT – tributyltin
TRV – toxicity reference value

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Table 4-49. Juvenile Chinook Salmon HQs Based on the Dietary Dose Line of Evidence

Round 2 COPC	Unit (bw/day)	IR Dose	NOAEL TRV	LOAEL TRV	NOAEL HQ	LOAEL HQ
Scenario 1[a]						
Cadmium	mg/kg	0.00159	0.0020	0.010	0.79	0.16
Copper	mg/kg	0.256	0.24	0.48	**1.1**	0.53
TBT	µg/kg	22.6	2.1	20	**11**	**1.1**
Total PCBs	µg/kg	16.4	10	50	**1.6**	0.33
Scenario 2[b]						
Total PCBs	µg/kg	6.21	10	50	0.62	0.12

[a] Scenario 1 represents the dietary dose based on the modeled juvenile chinook salmon diet.

[b] Scenario 2 represented the dietary dose based on juvenile chinook salmon stomach contents. Metals and butyltins were not analyzed in this scenario.

bw – body weight
COPC – chemical of potential concern
HQ – hazard quotient
IR – ingestion rate
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TBT – tributyltin
TRV – toxicity reference value
Bold identifies HQs greater than 1.0.

Table 4-50. Smallmouth Bass HQs Based on the Dietary Dose Line of Evidence

Round 2 COPC	Unit (bw/day)	IR Dose	NOAEL TRV	LOAEL TRV	NOAEL HQ	LOAEL HQ
Cadmium	mg/kg	0.000702	0.0020	0.010	0.35	0.070
Copper	mg/kg	0.0952	0.24	0.48	0.40	0.20
TBT	µg/kg	22.5	2.1	20	**11**	**1.1**
Total PCBs	µg/kg	54.1	10	50	**5.4**	**1.1**
Total DDTs	µg/kg	53.9	28	140	**1.9**	0.38

bw – body weight
COPC – chemical of potential concern
HQ – hazard quotient
IR – ingestion rate
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TBT – tributyltin
TRV – toxicity reference value
Bold identifies HQs greater than 1.0.

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Table 4-51. Northern Pikeminnow HQs Based on the Dietary Dose Line of Evidence

Round 2 COPC	Unit (bw/day)	IR Dose	NOAEL TRV	LOAEL TRV	NOAEL HQ	LOAEL HQ
Cadmium	mg/kg	0.00167	0.0020	0.010	0.83	0.17
Copper	mg/kg	0.258	0.24	0.48	**1.1**	0.54
Mercury	mg/kg	0.00215	0.013	0.048	0.17	0.045
TBT	µg/kg	22.6	2.1	20	**11**	**1.1**
Total PCBs	µg/kg	60.8	10	50	**6.1**	**1.2**
Total DDTs	µg/kg	25.0	28	140	0.89	0.18

bw – body weight
COPC – chemical of potential concern
HQ – hazard quotient
IR – ingestion rate
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TBT – tributyltin
TRV – toxicity reference value
Bold identifies HQs greater than 1.0.

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Table 4-52. Surface Water HQs

COI	Unit	UCL EPC	Acute Eco SL	Acute HQ	Chronic Eco SL	Chronic HQ
Peristaltic Method						
Zinc, dissolved	mg/L	0.0049	0.0362	0.14	0.0365	0.13
Benzo(a)anthracene	µg/L	0.011	0.49	0.022	0.027	0.41
Benzo(a)pyrene	µg/L	0.013	0.24	0.054	0.014	0.93
4-Chloro-3-methylphenol	µg/L	0.1	30	0.0033	0.6	0.17
Total PCBs	µg/L	0.0051	2	0.0026	0.014	0.36
2,4′-DDD	µg/L	0.00025	1.1	0.00023	0.001	0.25
2,4′-DDT	µg/L	0.0021	1.1	0.0019	0.001	**2.1**
4,4′-DDD	µg/L	0.00028	1.1	0.00025	0.001	0.28
4,4′-DDT	µg/L	0.00031	1.1	0.00028	0.001	0.31
Total DDTs	µg/L	0.0024	1.1	0.0022	0.001	**2.4**
XAD Method						
Benzo(a)anthracene	µg/L	0.0017	0.49	0.0035	0.027	0.063
Benzo(a)pyrene	µg/L	0.00167	0.24	0.00696	0.014	0.12
Total PCBs	µg/L	0.00325	2	0.00163	0.014	0.23
2,4′-DDD	µg/L	0.0014	1.1	0.0013	0.001	**1.4**
2,4′-DDT	µg/L	0.00031	1.1	0.00028	0.001	0.31
4,4′-DDD	µg/L	0.0027	1.1	0.0025	0.001	**2.7**
4,4′-DDT	µg/L	0.00212	1.1	0.00193	0.001	**2.1**
Total DDTs	µg/L	0.00672	1.1	0.00611	0.001	**6.7**

COI – chemical of interest
Eco SL – ecological screening level
EPC – exposure point concentration
HQ – hazard quotient
UCL – upper confidence limit

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Table 4-53. COIs for Which Non-detected Concentrations Exceeded SL TRVs

Receptor	COI	Non-Detected Concentration			Detected Concentration		
		Min RL	SL TRV HQ	Max RL	SL TRV HQ	Max Detect	SL TRV HQ
Largescale sucker	beta-HCH	3.3 UJ	0.67	8.5 UT	**1.7**	ND	ND
	delta-HCH	1.0 U	0.20	7.3 UT	**1.5**	ND	ND
Sculpin	dibutyl phthalate	270 U	1.0	330 U	**1.2**	ND	ND
	delta-HCH	1.0 UJ	0.20	9.6 U	**2.0**	1.6 NJ	0.33
	hexachlorobutadiene	1.0 UJT	0.038	33 UT	**1.3**	2 JT	0.077
Juvenile chinook salmon	BEHP	95 UT	0.24	860 UT	**2.2**	ND	ND
	butylbenzyl phthalate	190 U	0.16	3,700 U	**3.1**	220	0.18
	dibutyl phthalate	31 U	0.11	520 U	**1.9**	48	0.18
Smallmouth bass	beta-HCH	1.2 UJT	0.24	8.5 UT	**1.7**	ND	ND
	delta-HCH	1.0 UT	0.20	7.3 UT	**1.5**	ND	ND
	dibutyl phthalate	250 U	0.93	330 U	**1.2**	ND	ND
Northern pikeminnow	beta-HCH	4.0 UT	0.82	9.8 U	**2.0**	ND	ND
	delta-HCH	4.0 UT	0.82	9.8 U	**2.0**	ND	ND

BEHP – bis(2-ethylhexyl) phthalate
COI – chemical of interest
HCH – hexachlorocyclohexane
HQ – hazard quotient
J – estimated concentration
SL – screening level
N – presumptive evidence of a compound
ND – not detected
RL – reporting limit
T – value calculated or selected from multiple results
TRV – toxicity reference value
U – not detected at given concentration
Bold identifies HQs greater than 1.0.

Table 4-54. Comparison of 2,3,7,8-TCDD and TEQ Concentrations to SL TRVs for 2,3,7,8-TCDD

COI	Maximum Concentration in Tissue (pg/g)	SL TRV (pg/g)	Exceeds SL TRV?
Carp			
2,3,7,8-TCDD	1.07	90	no
PCB TEQ	1.35	90	no
Dioxin TEQ	10.8	90	no
Sculpin			
2,3,7,8-TCDD	0.446 T	90	no
PCB TEQ	2.11	90	no
Dioxin TEQ	32.3	90	no
Juvenile chinook salmon			
2,3,7,8-TCDD	0.876	90	no
PCB TEQ	0.225	90	no
Dioxin TEQ	4.32	90	no
Smallmouth bass			
2,3,7,8-TCDD	1.49	90	no
PCB TEQ	1.14	90	no
Dioxin TEQ	9.84	90	no

SL – screening level
HQ – hazard quotient
PCB – polychlorinated biphenyl
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
TRV – toxicity reference value

Table 4-55. Results of Dietary Portion Uncertainty Screen

Round 2 COPC	Receptor	Reason for Further Evaluation	Prey Item with Maximum Concentration	Assigned Percentage of Diet	Approximate Percentage of Diet Required for Exceedance	Likelihood of Occurrence
Copper	smallmouth bass	new NOAEL and LOAEL exceedance	crayfish	5%	crayfish ≥ 31% (for new NOAEL) and ≥ 73% (for new LOAEL)	moderate
	northern pikeminnow	new LOAEL exceedance	crayfish	30%	crayfish ≥ 69%	low
	pre-breeding sturgeon[a]	new NOAEL exceedance	clam	12%	clam ≥ 19%	high
Mercury	northern pikeminnow	new NOAEL exceedance	northern pikeminnow	5%	northern pikeminnow ≥ 75%	low
Total PCBs	northern pikeminnow	NOAEL and LOAEL doubled in magnitude	carp	5%	carp ≥ 29%	low
	peamouth	NOAEL doubled in magnitude and new LOAEL exceedance	worm	25%	worm ≥ 78% (for NOAEL to double) worm ≥ 34% (for new LOAEL)	low to moderate
Total DDTs	sculpin	new NOAEL exceedance	sculpin	20%	sculpin ≥ 28%	moderate
	peamouth	new NOAEL exceedance	sculpin	10%	sculpin ≥ 34%	low
	northern pikeminnow	new NOAEL exceedance	sculpin	25%	sculpin ≥ 31%	high

[a] Based on Scenario 1 where pre-breeding sturgeon where 8% incidental sediment ingestion is assumed.

COPC – chemical of potential concern

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

Table 4-56. Results of the Round 2 COPC/Receptor Pair Sediment Ingestion Rate Uncertainty Evaluation

Receptor	Round 2 COPC	Assumed SI (%)	Change in Risk Results	Likelihood of Occurrence
Largescale sucker	copper	8%	new LOAEL exceedance if SI ≥ 46%	Moderate; largescale suckers are benthic fish feeding in sediment.
Pre-breeding white sturgeon	copper	8%	new LOAEL exceedance if SI ≥ 69%	Moderate; sturgeon are benthic fish feeding in sediment.
Juvenile chinook salmon	copper	1%	new LOAEL exceedance if SI ≥ 39%	Low; juvenile chinook salmonids are pelagic feeders.
Sculpin	copper	5%	new LOAEL exceedance if SI ≥ 41%	Low; sculpin are benthic fish with some exposure to sediments.
	TBT	5%	new LOAEL exceedance if SI ≥ 69%	
Peamouth	copper	5%	new LOAEL exceedance if SI ≥ 39%	Low; peamouth are benthopelagic fish with low exposure to sediments.
	TBT	5%	new LOAEL exceedance if SI ≥ 19%	
Smallmouth bass	copper	1%	new LOAEL exceedance if SI ≥ 54%	Low; smallmouth bass are pelagic feeders with low exposure to sediments.
	cadmium	1%	new NOAEL exceedance if SI ≥ 5%	Moderate; smallmouth bass are pelagic feeders with low exposure to sediments.
Northern pikeminnow	copper	1%	new LOAEL exceedance if SI ≥ 32%	Low to moderate; northern pikeminnow are pelagic feeders with low exposure to sediments.
	cadmium	1%	new NOAEL exceedance if SI ≥ 5.5%	
	total DDTs	1%	new NOAEL exceedance if SI ≥ 83%	

COPC – chemical of potential concern
SI – sediment ingestion
TBT -- tributyltin

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Table 4-57. Comparison of HQs for Fish Receptors Based on UCL EPCs and Mean EPCs

Round 2 COPC	HQs Based on UCL EPCs		HQs Based on Mean EPCs	
	NOAEL	LOAEL	NOAEL	LOAEL
Sculpin				
Cadmium	**1.8**	0.36	**1.5**	0.30
Copper	**1.1**	0.56	**1.0**	0.51
Tributyltin ion	**8.3**	0.87	0.98	0.10
Total PCBs	**6.1**	**1.2**	**1.9**	0.39
Peamouth				
Cadmium	**1.5**	0.30	**1.2**	0.25
Copper	**1.2**	0.58	0.98	0.49
Tributyltin ion	**9.3**	0.97	**1.1**	0.11
Total PCBs	**4.3**	0.86	**1.3**	0.27
Juvenile chinook salmon				
Copper	**1.1**	0.53	0.80	0.40
Tributyltin ion	**11**	**1.1**	**1.1**	0.11
Total PCBs	**1.6**	0.33	0.47	0.094
Largescale sucker				
Cadmium	**2.2**	0.44	**1.8**	0.36
Copper	**1.3**	0.67	**1.2**	0.60
Tributyltin ion	**7.7**	0.81	0.95	0.10
Total PCBs	**6.2**	**1.2**	**1.8**	0.36
Smallmouth bass				
Tributyltin ion	**11**	**1.1**	**1.1**	0.11
Total PCBs	**5.4**	**1.1**	**2.4**	0.48
Total DDTs	**1.9**	0.38	0.31	0.061
Northern pikeminnow				
Copper	**1.1**	0.54	0.97	0.48
Tributyltin ion	**11**	**1.1**	**1.1**	0.11
Total PCBs	**6.1**	**1.2**	**2.1**	0.42
Pre-breeding sturgeon – 8% SIR				
Cadmium	**2.0**	0.39	**1.6**	0.32
Tributyltin ion	**10**	**1.1**	**1.1**	0.12
Total PCBs	**9.1**	**1.8**	**2.4**	0.48
Pre-breeding sturgeon – 56% SIR				
Cadmium	**3.0**	0.60	**2.2**	0.44

Table 4-57. Comparison of HQs for Fish Receptors Based on UCL EPCs and Mean EPCs

Round 2 COPC	HQs Based on UCL EPCs		HQs Based on Mean EPCs	
	NOAEL	LOAEL	NOAEL	LOAEL
Copper	**1.8**	0.88	**1.5**	0.74
Tributyltin ion	**11**	**1.1**	**2.0**	0.21
Total PCBs	**9.2**	**1.8**	**2.4**	0.49

COPC – chemical of potential concern
EPC – exposure point concentration
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
SIR – sediment ingestion rate
UCL – upper confidence limit

Table 4-58. Summary Uncertainties for Metals Round 2 COPCs in Fish

Line of Evidence	Receptor	NOAEL or Chronic HQ	LOAEL or Acute HQ	Exposure Uncertainties	Effects Uncertainties
Cadmium					
Tissue residue	all fish	Not a Round 2 COPC; no risks expected based on this LOE.			
Dietary dose	largescale sucker	**2.2**	0.44	Low: Changing dietary portions, sediment ingestion, and/or exposure scale does not result in significantly higher HQs.	Moderate: Selected LOAEL TRV is several orders of magnitude lower than other toxicological studies and may overestimate toxicity of cadmium; NOAEL TRV based on UF.
	pre-breeding sturgeon[a]	**2.0**; **3.0**	0.39; 0.60		
	sculpin	**1.8**	0.36	Moderate: Reducing exposure scale to less than site-wide may result in LOAEL HQs > 1.0 in localized areas.	
	peamouth	**1.5**	0.30	Low: changing dietary portions, sediment ingestion, and/or exposure scale does not result in significantly higher HQs	
	juvenile chinook salmon	0.7980	0.16		
	smallmouth bass	0.35	0.070	Low to moderate: Slightly increasing SI (to > 5%) would result in NOAEL HQ > 1.0 and reducing exposure scale to less than site-wide may result in NOAEL HQs > 1.0 in localized areas; however, LOAEL HQs would still be < 1.0.	
	northern pikeminnow	0.83	0.17		
Surface water	all fish	Not a Round 2 COPC; no risks expected based on this LOE.			
TZW	sculpin/lamprey ammocoetes	Not a potential iCOC; no risks expected based on this LOE.			
Chromium					
Tissue residue	largescale sucker	0.35[b] (0.069 – 0.50)	0.22[b] (0.044–0.32)	High: Fish actively regulate lead and exposure concentrations vary depending on time of exposure.	High: Tissue TRV may not reflect critical tissue concentration because chromium is regulated by fish; limited toxicity data; toxicity study did not report tissue concentrations in a control group.
Copper					
Tissue residue	all fish	Not a Round 2 COPC; no risks expected based on this LOE.			

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Table 4-58. Summary Uncertainties for Metals Round 2 COPCs in Fish

Line of Evidence	Receptor	NOAEL or Chronic HQ	LOAEL or Acute HQ	Exposure Uncertainties	Effects Uncertainties
Dietary dose	largescale sucker	**1.3**	0.67	Low: A large increase in SI would result in LOAEL HQ > 1.0; however, likelihood of occurrence is low.	Low: Lowest toxicological thresholds reported in reviewed literature.
	pre-breeding sturgeon[a]	0.93; **1.8**	0.46; 0.88	High: Higher consumption of worms and increasing SI would result in LOAEL HQ > 1.0.	
	sculpin	**1.1**	0.56	Moderate: Reducing exposure scale to less than site-wide may result in LOAEL HQs > 1.0 in localized areas; and a large increase in SI would result in LOAEL HQ > 1.0; however, likelihood of increased SI is low.	
	peamouth	**1.2**	0.58	Low: A large increase in SI would result in LOAEL HQ > 1.0; however, likelihood of occurrence is low.	
	juvenile chinook salmon	**1.1**	0.53		
	smallmouth bass	0.40	0.20	Moderate: Higher consumption of crayfish would result in LOAEL HQ > 1.0; a large increase SI would result in LOAEL HQ > 1.0; however, likelihood of occurrence of is moderate.	
	northern pikeminnow	**1.1**	0.54	Low: Higher consumption of crayfish and a large increase SI would result in LOAEL HQ > 1.0; however, likelihood of occurrence of is low	
Surface water	all fish	Not a Round 2 COPC; no risks expected based on this LOE.			
TZW	sculpin/lamprey ammocoetes	Not a potential iCOC; no risks expected based on this LOE.			
Lead					
Tissue residue	peamouth	**15**[b] (0.012 – **4.2**)	**9.2**[b] (0.077 – **2.6**)	High: Fish actively regulate lead, and exposure concentrations vary depending on time of exposure.	Moderate: Tissue TRV may not reflect critical tissue concentration because lead is regulated by fish.
	all other fish	Not a Round 2 COPC; no risks expected based on this LOE.			
Dietary dose	all fish	Not a Round 2 COPC; no risks expected based on this LOE.			
Surface water	all fish	Not a Round 2 COPC; no risks expected based on this LOE.			
TZW	sculpin/lamprey ammocoetes	Not a potential iCOC; no risks expected based on this LOE.			

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Table 4-58. Summary Uncertainties for Metals Round 2 COPCs in Fish

Line of Evidence	Receptor	NOAEL or Chronic HQ	LOAEL or Acute HQ	Exposure Uncertainties	Effects Uncertainties
Mercury					
Tissue residue	northern pikeminnow	**2.0**[b] (0.73 – **2.5**)	0.85[b] (0.31 – **1.1**)	One tissue composite (07R009) has a LOAEL HQ of 1.1; however, UCL LOAEL HQ is 0.85.	Low: Lowest toxicological thresholds reported in reviewed literature based on direct measure of survival, growth, or reproduction.
	all other fish	Not a Round 2 COPC; no risks expected based on this LOE.			
Dietary dose	pre-breeding sturgeon[a]	0.025; 0.053	0.0069; 0.014	Low: Changing dietary portions, sediment ingestion, and/or exposure scale does not result in significantly higher HQs.	Low: Lowest toxicological thresholds reported in reviewed literature based on direct measure of survival, growth, or reproduction.
	northern pikeminnow	0.17	0.045	Low: Higher consumption of pikeminnow would result in NOAEL HQ > 1.0; however, likelihood of occurrence is low.	
	all other fish	Not a Round 2 COPC; no risks expected based on this LOE.			
Surface water	all fish	Not a Round 2 COPC; no risks expected based on this LOE.			
TZW	sculpin/lamprey ammocoetes	Not a potential iCOC; no risks expected based on this LOE.			
Zinc					
Tissue residue	juvenile chinook salmon	0.90[b] (0.71 – 0.98)	0.68[b] (0.60 – 0.83)	High: Fish actively regulate zinc and exposure concentrations vary depending on time of exposure.	Moderate: Tissue TRV may not reflect critical tissue concentration because lead is regulated by fish.
	all other fish	Not a Round 2 COPC; no risks expected based on this LOE.			
Dietary dose	all fish	Not a Round 2 COPC; no risks expected based on this LOE.			
Surface water	all fish	0.13	0.14	Low: Low volume sampling method of peristaltic method may over or underestimate risk.	Low: Eco SLs expected to be protected of fish receptors.
TZW	sculpin/lamprey ammocoetes	Not a potential iCOC; no risks expected based on this LOE.			

[a] Two scenarios were evaluated for pre-breeding sturgeon where sturgeon were assumed to consume 8% and 56% sediment; the range of NOAEL and LOAEL HQs from both scenarios are presented.

[b] HQs based on UCL EPCs. Range of NOAEL and LOAEL HQs are presented for the critical tissue-residue LOE based on individual composite locations are in parentheses.

COPC – chemical of potential concern
Eco SL – ecological screening level
HQ – hazard quotient
iCOC – initial chemical of concern
LOAEL – lowest-observed-adverse-effect level
Bold identifies HQs greater than 1.0.
LOE – line of evidence
NOAEL – no-observed-adverse-effect level
TRV – toxicity reference value
TZW – transition zone waterUF – uncertainty factor

BZTO104(e)030849

Table 4-59. Summary of Key Uncertainties for TBT

LOE	Receptor	NOAEL or Chronic HQ	LOAEL or Acute HQ	Exposure Uncertainties	Effects Uncertainties
Tissue residue	all fish	Not a Round 2 COPC; no risks expected based on this LOE.		High: LOAEL HQ > 1.0 assuming laboratory-tested worm or field-collected clam tissue to represent TBT concentrations in fish tissue; however, fish are likely to have lower TBT tissue concentrations than their prey.	Moderate: Limited toxicological studies available (n=3).
Dietary dose	largescale sucker	**7.7**	0.81	Low: Diet is represented using 100% benthic invertebrates.	Moderate: Limited toxicological studies available (n=1).
	pre-breeding sturgeon[a]	**10**; **11**	**1.1**; **1.1**		
	sculpin	**8.3**	0.87	High: Prey tissue data represented by clams and worms only; reducing exposure scale to less than site-wide may result in LOAEL HQs > 1.0 in localized areas; an increase in SI would result in LOAEL HQ > 1.0; however, likelihood of increased SI of is low.	
	peamouth	**9.3**	1.0	High: Prey tissue data represented by clams and worms only; an increase in SI would result in LOAEL HQ > 1.0; however, likelihood of increased SI of is low.	
	juvenile chinook salmon	**11**	**1.1**	High: Prey tissue data represented by worms only.	
	smallmouth bass	**11**	**1.1**		
	northern pikeminnow	**11**	**1.1**		
Surface water	all fish	Not a Round 2 COPC; no risks expected based on this LOE.			
TZW	sculpin/lamprey ammocoetes	Not a Round 2 COPC; no risks expected based on this LOE.			

[a] Two scenarios were evaluated for pre-breeding sturgeon where sturgeon were assumed to consume 8% and 56% sediment; the range of NOAEL and LOAEL HQs from both scenarios are presented.

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LOE – line of evidence
NOAEL – no-observed-adverse-effect level
TBT – tributyltin
TZW – transition zone water

Bold identifies HQs greater than 1.0.

BZTO104(e)030850

Table 4-60. Summary of Key Uncertainties for PAH Round 2 COPCs

Line of Evidence	Receptor	NOAEL or Chronic HQ	LOAEL or Acute HQ	Exposure Uncertainties	Effects Uncertainties
Benzo(a)anthracene					
Surface water[a]	all fish	0.41; 0.063	0.022; 0.0035	Low: Both XAD (high volume) and peristaltic (low volume) methods used.	Low: Eco SL expected to be protected of fish receptors.
TZW	sculpin/lamprey ammocoetes	0.089 – **1,200**	0.0049 – **66**	Individual locations of chronic exceedances were located at Arco, Kinder Morgan, Gasco, Siltronic, and ExxonMobil Oil.	
Benzo(a)pyrene					
Dietary dose	pre-breeding sturgeon[b]	0.039; 0.060	0.018; 0.028	Low: Changing dietary portions and/or sediment ingestion do not result in significantly higher HQs.	Low: Limited toxicological studies available.
	all other fish	Not a Round 2 COPC; no risks expected based on this LOE.			
Surface water[a]	all fish	0.12; 0.93	0.00696; 0.054	Low: Both XAD (high volume) and peristaltic (low volume) methods used.	Low: Eco SL expected to be protected of fish receptors.
TZW	sculpin/lamprey ammocoetes	0.13 – **2,700**	0.0075 – **160**	Individual locations of chronic exceedances were located at Arco, Gasco, Siltronic, and ExxonMobil Oil.	
Other individual PAHs					
TZW	sculpin/lamprey ammocoetes	0.00010 – **1,200**	0.00010 – **72**	Individual locations of chronic exceedances of other individual PAHs were located at Gasco and Siltronic properties; three additional PAHs (i.e., benzo(g,h,i)perylene, fluorene, and phenanthrene) exceeded chronic Eco SLs at ExxonMobil Oil.	Low: Eco SL expected to be protected of fish receptors.
Total PAHs					
Dietary dose	pre-breeding sturgeon[a]	0.10; 0.14[d]	0.035; 0.048[d]	Low: Changing dietary portions and/or sediment ingestion does not result in significantly higher HQs.	Moderate: Dietary-dose TRV based on dietary concentration of PAH mixture that includes some of the same PAHs included the ERA Total PAH sum, as well as additional PAHs; limited toxicological studies available; use of effects level threshold based on field toxicity data would result in LOAEL exceedances for several fish receptors.
	all other fish	Not a Round 2 COPC; no risks expected based on this LOE.			

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Table 4-60. Summary of Key Uncertainties for PAH Round 2 COPCs

Line of Evidence	Receptor	NOAEL or Chronic HQ	LOAEL or Acute HQ	Exposure Uncertainties	Effects Uncertainties
Surface water[b]	all fish	Not a Round 2 COPC; no risks expected based on this LOE.			
TZW	sculpin/lamprey ammocoetes	Not a Round 2 COPC; no risks expected based on this LOE.			

[a] Two scenarios were evaluated for pre-breeding sturgeon where sturgeon were assumed to consume 8% and 56% sediment; the range of NOAEL and LOAEL HQs from both scenarios are presented.

[b] A peristaltic pump method and XAD method were used to collect surface water; the range of acute and chronic HQs from both methods are presented.

COPC – chemical of potential concern
Eco SL – ecological screening level
ERA – ecological risk assessment
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LOE – line of evidence
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
TRV – toxicity reference value
TZW – transition zone water

Bold identifies HQs greater than 1.0.

BZTO104(e)030852

Table 4-61. Summary of Key Uncertainties for BEHP, 4-Chloro-3-methyl Phenol, and Cyanide

Line of Evidence	Receptor	NOAEL or Chronic HQ	LOAEL or Acute HQ	Exposure Uncertainties	Effects Uncertainties
4-Chloro-3-methylphenol					
Surface water	all fish	0.17	0.0033	Low: Low volume sampling method of peristaltic method may over or underestimate risk.	High: Chronic Eco SL derived from acute Eco SL divided by 50 and may overestimate risk to fish.
TZW	sculpin/lamprey ammocoetes	Not a Round 2 COPC; no risks expected based on this LOE.			
BEHP					
Tissue residue	largescale sucker	**14**[a] (0.20 – **7.7**)	0.52[a] (0.0074 – 0.28)	Moderate: Low detection frequency (33%) in tissue samples.	High: LOAEL TRV based derived using an egg-to-adult conversion factor based on PCB data; Limited toxicological data available (no other LOAEL TRV for BEHP reported in the literature).
	sculpin	**33**[a] (0.21 – **72**)	**1.2**[a] (0.0077 – **2.6**)	Moderate: Low detection frequency (12%) in tissue samples; one tissue composite (08R003) has a LOAEL HQ of 2.6; all other tissue composites had LOAEL HQs < 1.0.	
	smallmouth bass	**190**[a] (0.22 – **220**)	**6.9**[a] (0.0081 – **8.2**)	Moderate: Low detection frequency (14%) in tissue samples; two tissue composites (both at 04R023) have a LOAEL HQ > 1.0; all other tissue composites had LOAEL HQs < 1.0.	
	all other fish	Not a Round 2 COPC; no risks expected based on this LOE.			
Surface water	all fish	Not a Round 2 COPC; no risks expected based on this LOE.			
TZW	sculpin/lamprey ammocoetes	Not a Round 2 COPC; no risks expected based on this LOE.			
Cyanide					
TZW	sculpin/lamprey ammocoetes	**1.2 – 4,400**	0.27 – **1,100**	Individual locations of chronic Eco SL exceedances were located at Gasco and Siltronic.	Low: Eco SL expected to be protected of fish receptors.
Perchlorate					
TZW	sculpin/lamprey ammocoetes	NA	0.22 – **9,800**	Individual locations of chronic Eco SL exceedances were located at Arkema acid and chlordate plants.	Low: Eco SL expected to be protected of fish receptors.

[a] HQs based on UCL EPCs. Range of NOAEL and LOAEL HQs are presented for the critical tissue-residue LOE based on individual composite locations are in parentheses.

BEHP – bis(ethylhexyl) phthalate
COPC – chemical of potential concern
Eco SL – ecological screening level
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LOE – line of evidence
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TRV – toxicity reference value
TZW – transition zone water
UF – uncertainty factor

Bold identifies HQs that are greater than 1.0.

BZTO104(e)030853

Table 4-62. Summary of Key Uncertainties for Total PCBs

Line of Evidence	Receptor	NOAEL or Chronic HQ	LOAEL of Acute HQ	Exposure Uncertainties	Effects Uncertainties
Tissue residue	largescale sucker	0.81[a] (0.051 – **1.1**)	0.38[a] (0.024 – 0.51)	No individual tissue composites have LOAEL HQs > 1.0.	High: TRVs based on lowest appropriate whole-body tissue concentration reported in toxicological studies; however based on egg exposure to high PCB water concentrations; LOAEL derived using egg-to-adult conversion factor.
	sculpin	0.68[a] (0.053 – **1.8**)	0.32[a] (0.025 – 0.85)		
	smallmouth bass	1.0[a] (0.057 – **2.6**)	0.49[a] (0.027 – **1.2**)	One tissue composite (08R010) has a LOAEL HQ of 1.2; all other tissue composites had LOAEL HQs < 1.0.	
	northern pikeminnow	0.71[a] (0.20 – 1.0)	0.34[a] (0.094 – 0.48)	No individual tissue composites have LOAEL HQs > 1.0.	
	all other fish	Not a Round 2 COPC; no risks expected based on this LOE.			
Dietary dose	largescale sucker	**6.2**	**1.2**	Low: Changing dietary portions, sediment ingestion, and/or exposure scale does not result in significantly higher HQs.	Low: TRVs based on lowest appropriate dietary dose estimates reported in toxicological studies; NOAEL derived using a UF.
	pre-breeding sturgeon	**9.1; 9.2**	**1.8**		
	sculpin	**6.1**	**1.2**		
	peamouth	**4.3**	0.86	Moderate: Higher consumption of worms would result in LOAEL HQ > 1.0.	
	juvenile chinook salmon[b]	**1.6**; 0.62	0.33; 0.12	Low: Changing dietary portions, sediment ingestion, and/or exposure scale does not result in significantly higher HQs.	
	smallmouth bass	**5.4**	**1.1**		
	northern pikeminnow	**6.1**	**1.2**	Low: Higher consumption of carp would increase magnitude of HQs; however, likelihood of occurrence of is low.	
Surface water[c]	all fish	0.36; 0.23	0.0024; 0.00163	Low: Both XAD (high volume) and peristaltic (low volume) methods used.	Low: Eco SL expected to be protected of fish receptors.
TZW	sculpin/lamprey ammocoetes	Not a Round 2 COPC; no risks expected based for this LOE.			

[a] HQs based on UCL EPCs. Range of NOAEL and LOAEL HQs are presented for the critical tissue-residue LOE based on individual composite locations are in parentheses.

[b] Two dietary dose scenarios were evaluated for juvenile chinook salmon based on either the modeled chinook salmon diet or juvenile chinook stomach contents; the range of NOAEL and LOAEL HQs from both scenarios are presented.

[c] A peristaltic pump method and XAD method were used to collect surface water; the range of acute and chronic HQs from both methods are presented.

HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TRV – toxicity reference value
TZW – transition zone water
UF – uncertainty factor

Bold identifies HQs that are greater than 1.0.

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Table 4-63. Summary of Key Uncertainties for the Pesticide Round 2 COPCs

Line of Evidence	Receptor	NOAEL or Chronic HQ	LOAEL of Acute HQ	Exposure uncertainties	Effects uncertainties
beta-HCH					
Tissue residue	sculpin	0.0030 (0.00063 – 0.0061)[a]	0.000061 (0.000013 – 0.000122)[a]	No individual tissue composites have HQs > 1.0.	High: Use of surrogate TRV (based on gamma-HCH) may over or underestimate toxicity; TRVs extrapolated from LC50; limited toxicological data available.
	all other fish	Not a Round 2 COPC; no risks expected based on this LOE.			
Surface water	all fish	Not a Round 2 COPC; no risks expected based on this LOE.			
TZW	sculpin/lamprey ammocoetes	Not a Round 2 COPC; no risks expected based on this LOE.			
2,4′-DDD					
Surface water[b]	all fish	0.25; **1.4**	0.00023; 0.0013	Low: Both XAD (high volume) and peristaltic (low volume) methods used.	Low: Eco SL expected to be protected of fish receptors.
TZW	sculpin/lamprey ammocoetes	**3.3 – 1,100**	0.003 – **1.0**	Individual locations of chronic exceedances were located Arkema (acid plant) and Rhône-Poulenc.	
2,4′-DDT					
Surface water[b]	all fish	**2.1**; 0.31	0.0019; 0.00028	High (for peristaltic data): Exposure concentrations based on NJ-qualified data.	Low: Eco SL expected to be protected of fish receptors.
TZW	sculpin/lamprey ammocoetes	0.89 – **150**	0.00081 – 0.14	Individual locations of chronic exceedances were located Arkema (acid plant) and Rhône-Poulenc.	
4,4′-DDD					
Surface water[b]	all fish	0.28; **2.7**	0.00025; 0.0025	Low: Both XAD (high volume) and peristaltic (low volume) methods used.	Low: Eco SL expected to be protected of fish receptors.
TZW	sculpin/lamprey ammocoetes	**4.7 – 1,300**	0.0043 – **1.2**	Individual locations of chronic exceedances were located Arkema (acid plant) and Rhône-Poulenc.	
4,4′-DDE					
TZW	sculpin/lamprey ammocoetes	**3.9 – 930**	0.0035 – 0.85	Individual locations of chronic exceedances were located Arkema (acid plant) and Rhône-Poulenc.	Low: Eco SL expected to be protected of fish receptors.

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Table 4-63. Summary of Key Uncertainties for the Pesticide Round 2 COPCs

Line of Evidence	Receptor	NOAEL or Chronic HQ	LOAEL of Acute HQ	Exposure uncertainties	Effects uncertainties
4,4′-DDT					
Surface water[b]	all fish	0.31; **2.1**	0.00028; 0.00193	Low: Both XAD (high volume) and peristaltic (low volume) methods used.	Low: Eco SL expected to be protected of fish receptors.
TZW	sculpin/lamprey ammocoetes	**5 – 1,800**	0.0045 – **1.6**	Individual locations of chronic exceedances were located Arkema (acid plant).	
Total DDTs					
Tissue residue	largescale sucker	0.35 (0.084 – 0.37)[a]	0.35 (0.084 – 0.37)[a]	No individual tissue composites have HQs > 1.0.	Low: TRVs based on lowest appropriate critical tissue-residue concentrations reported in toxicological studies.
	sculpin	0.78 (0.018 – **1.7**)[a]	0.78 (0.018 – **1.7**)[a]	One tissue composite (07R006) has a LOAEL HQ of 1.7; all other tissue composites had LOAEL HQs < 1.0.	
	smallmouth bass	0.15 (0.044 – 0.23)[a]	0.15 (0.044 – 0.23)[a]	No individual tissue composites have HQs > 1.0.	
	northern pikeminnow	0.33 (0.091 – 0.42)[a]	0.33 (0.091 – 0.42)[a]	No individual tissue composites have HQs > 1.0.	
	all other fish	Not a Round 2 COPC; no risks expected based on this LOE.			
Dietary dose	largescale sucker	0.56	0.11	Low: Changing dietary portions, sediment ingestion, and/or exposure scale does not result in significantly higher HQs.	Low: TRVs based on lowest appropriate dietary dose estimates reported in toxicological studies; NOAEL derived using a UF.
	pre-breeding sturgeon[c]	0.83; 0.87	0.17		
	sculpin	0.87	0.17	Moderate: Reducing exposure scale to less than site-wide may result in NOAEL HQs > 1.0 in localized areas; however, LOAEL HQs would still be < 1.0.	
	peamouth	0.53	0.11	Moderate: higher consumption of sculpin would result in NOAEL > 1.0.	
	smallmouth bass	**1.9**	0.38	Low: Changing dietary portions, sediment ingestion, and/or exposure scale does not result in significantly higher HQs.	
	northern pikeminnow	0.89	0.18	High: higher consumption of sculpin, increase in SI, and reducing exposure scale to less than site-wide would result in NOAEL HQs > 1.0; however, LOAEL HQs would still be < 1.0.	

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Table 4-63. Summary of Key Uncertainties for the Pesticide Round 2 COPCs

Line of Evidence	Receptor	NOAEL or Chronic HQ	LOAEL of Acute HQ	Exposure uncertainties	Effects uncertainties
Surface water[b]	all fish	**2.4**; **6.7**	0.0022; 0.00611	High (for peristaltic data): Exposure concentrations based on NJ-qualified data.	Low: Eco SL expected to be protected of fish receptors.
TZW	sculpin/lamprey ammocoetes	**7.7 – 3,100**	0.007 – **2.8**	Individual locations of chronic exceedances were located Arkema (acid plant) and Rhône-Poulenc.	

[a] HQs based on UCL EPCs. Range of NOAEL and LOAEL HQs are presented for the critical tissue-residue LOE based on individual composite locations are in parentheses.

[b] A peristaltic pump method and XAD method were used to collect surface water; the range of acute and chronic HQs from both methods are presented.

[c] Two scenarios were evaluated for pre-breeding sturgeon where sturgeon were assumed to consume 8% and 56% sediment, respectively; the range of NOAEL and LOAEL HQs from both scenarios are presented.

COPC – chemical of potential concern
Eco SL – ecological screening level
HCH – hexachlorocyclohexane
HQ – hazard quotient
J – estimated concentration
LOAEL – lowest-observed-adverse-effect level
Bold identifies HQs that are greater than 1.0.
N – presumptive evidence of a compound
NOAEL – no-observed-adverse-effect level
TRV – toxicity reference value
TZW – transition zone water
UF – uncertainty factor

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Table 5-1. Resident Bird Species that Potentially Breed Along the LWR

Species	Scientific Name	Residency Status[a]
Herbivores		
Canada goose	*Branta canadensis*	some residents; some winter guests
Mallard	*Anas platyrhynchos*	year-round
Diving Carnivores and Omnivores		
American dipper	*Cinclus mexicanus*	mostly year-round
Common merganser	*Mergus merganser*	mostly year-round
Hooded merganser	*Lophodytes cucullatus*	some year-round; some winter guests
Pied-billed grebe	*Podilyumbus podiceps*	summer; many winter guests
American coot	*Fulica americana*	year-round
Cinnamon teal	*Anas cyanoptera*	summer
Wood duck	*Aix sponsa*	some year-round
Sediment-Probing Invertivores and Omnivores		
Common snipe	*Gallinago gallinago*	mostly year-round
Killdeer	*Charadrium vociferous*	year-round; some winter guests
Spotted sandpiper	*Actitis macularia*	some year-round
Sora	*Porzana carolina*	mostly summer; some winter guests
Virginia rail	*Rallus limicola*	some year-round
civores		
American bittern	*Botaurus lentiginosus*	year-round
Bald eagle[b]	*Haliaeetus leucocephalus*	mostly year-round; some winter guests
Belted kingfisher	*Ceryle alcyon*	mostly year-round
Double-crested cormorant	*Phalacrocorax auritus*	some summer; many winter guests
Great blue heron	*Ardea herodias*	year-round
Green heron	*Butorides virescens*	some year-round
Osprey	*Pandion halieatus*	summer

[a] Puchy and Marshall (1993).
[b] Federally and Oregon state-listed as threatened species.

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Table 5-2. Birds that are Seasonally or Minimally Associated with Aquatic Habitat in the Study Area

Species	Scientific Name
Carnivore	
American peregrine falcon[a]	*Falco peregrinus*
Diving Carnivore	
Barrow's goldeneye	*Bucephala islandica*
Bonaparte's gull	*Larus philadelphia*
Bufflehead	*Bucephala albeola*
Common goldeneye	*Bucephala clangula*
Common teal (green-winged teal)	*Anas carolinensis*
Eared grebe	*Podiceps nigricollis*
Greater scaup	*Aythya marila*
Harlequin duck[b]	*Histrionicus histrionicus*
Horned grebe	*Podiceps auritus*
Lesser scaup	*Aythya affinis*
Red-necked grebe[c]	*Podiceps grisegena*
Ruddy duck	*Oxyura jamaicensis*
Wilson's phalarope	*Phalaropus tricolor*
Diving Herbivore	
Aleutian Canada goose	*Branta canadensis leucopareia*
American wigeon	*Anas Americana*
Canvasback	*Aythya valisineria*
Gadwall	*Anas strepera*
Northern pintail	*Anas acuta*
Northern shoveler	*Anas clypeata*
Ring-necked duck	*Aythya collaris*
Tundra swan	*Cygnus columbianus*
Diving Piscivore	
Common Loon	*Gavia immer*
Pelagic cormorant	*Phalacrocorax pelagicus*
Western grebe	*Aechmophorus occidentalis*
Insectivore	
American pipit	*Anthus rubescens*
Barn swallow	*Hirundo rustica*
Black tern[b]	*Chlidonias niger*
Cliff swallow	*Hirundo pyrrhonota*
Omnivore	
Ring-billed gull	*Larus delawarensis*
Sandhill crane	*Grus Canadensis*
Tri-colored blackbird[b]	*Agelaius tricolor*
Piscivore	
American white pelican	*Pelecanus erythrorhynchos*

Table 5-2. Birds that are Seasonally or Minimally Associated with Aquatic Habitat in the Study Area

Species	Scientific Name
Black-crowned night heron	*Nycticorax nycticorax*
Caspian tern	*Sterna caspia*
Forster's tern	*Sterna forsteri*
Great egret	*Ardea alba*
Greater yellowlegs	*Tringa melanoleuca*
Western gull	*Larus occidentalis*
Sediment-Probing Carnivore	
California gull	*Larus californicus*
Dunlin	*Calidris alpine*
Greater yellowlegs	*Tringa melanoleuca*
Least sandpiper	*Calidris minutilla*
Lesser yellowlegs	*Tringa flavipes*
Long-billed curlew	*Numenius americanus*
Long-billed dowitcher	*Limnodromus scolopaceus*
Marbled godwit	*Limosa fedoa*
Mew gull	*Larus canus*
Semipalmated plover	*Charadrius semipalmatus*
Short-billed dowitcher	*Limnodromus griseus*
Solitary sandpiper	*Tringa solitaria*
Western sandpiper	*Calidris mauri*
Willet	*Catoptrophorus semipalmatus*

Source: Csuti et al. (2001)

[a] Oregon state listed as endangered.

[b] Federally listed as a species of concern.

[c] Listed as a critical species on the Oregon Department of Fish and Wildlife sensitive species list (Oregon Natural Heritage Program 2004).

Table 5-3. Mammals that Potentially Use the Lower Willamette River

Species	Scientific Name
Beaver	*Castor canadensis*
California sea lion	*Zalophus californianus*
Mink	*Mustela vision*
Muskrat	*Ondatra zibethicus*
Nutria	*Myocastor coypus*
Raccoon	*Procyon lotor*
River otter	*Lutra canadensis*

Source: Csuti et al. (2001)

Table 5-4. Lines of Evidence Used to Evaluate Risks to Wildlife Receptors in the LWR

Receptor of Concern	Assessment Endpoint	Measures of Effect and Exposure (Measurement Endpoints)	Lines of Evidence in Support of the Measurement Endpoints
Spotted sandpiper[a]	survival, growth, and reproduction	dietary dose (empirical or modeled via food chain or FWM) compared to dietary TRVs[b]	Dietary-based TRV approach. Sediment concentrations were determined from site-specific evaluation. In the absence of appropriate invertebrate tissue concentrations, modeled invertebrate tissue concentrations were used.
Hooded merganser	survival, growth, and reproduction	dietary dose (empirical or modeled via food chain or FWM) compared to dietary TRVs[b]	Dietary-based TRV approach. Dietary-based analysis used sculpin and/or invertebrate tissue data to represent feeding guild. In the absence of appropriate fish and invertebrate tissue concentrations, modeled concentrations were used. For dioxin-like contaminants (carp or appropriate prey species), a TEQ-based approach was used to assess reproductive effects.
Bald eagle	survival, growth, and reproduction	dietary dose (empirical or modeled via food chain or FWM) compared to dietary TRVs[b]	Dietary-based approach incorporated food chain transfer of contaminants from appropriate fish species (assuming all exposure comes from prey fish). Assessed dioxin-like contaminants using a TEQ approach based on appropriate surrogate fish tissue data. TRVs based on the most sensitive life stages were used. Water intake as component of dietary exposure models was not included in the quantitative risk assessments, but the impact of water ingestion on exposure calculations is considered in Section 5.4.2.
			Dietary-based approach included egg or embryo-based TRVs for DDT and metabolites, PCBs, and dioxin-like compounds. Egg concentrations were determined by egg analysis or by food chain modeling.
Osprey	survival, growth, and reproduction	dietary dose (empirical or modeled via food chain or FWM) compared to dietary TRVs[b]	Dietary-based approach incorporating food chain transfer of contaminants from appropriate fish species (assuming all exposure comes from prey fish). Dioxin-like contaminants were assessed using a TEQ approach based on appropriate surrogate fish tissue data. TRVs based on the most sensitive life stages were used. not included in the quantitative risk assessments, but the impact of water ingestion on exposure calculations was evaluated in Section 5.4.2
			Dietary-based approach included egg or embryo-based TRVs for DDT and metabolites, PCBs, and dioxin-like compounds. Egg concentrations were determined by egg analysis or by food chain modeling.
Mink and river otter	survival, growth, and reproduction	dietary dose (empirical or modeled via food chain or FWM) compared to dietary TRVs[b]	Dietary-based TRV approach. Considered both relevant fish species concentrations and invertebrate (crayfish) components of the diet. For dioxin-like contaminants (carp or appropriate prey species), a TEQ-based approach was used to assess reproductive effects.

[a] Considered representative of sediment ingestion.

[b] Dietary estimates for all wildlife receptors were based on empirical data.

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Table 5-5. Round 2 COPC/Receptor Pairs Evaluated for Wildlife Receptors

Round 2 COPC	Wildlife Receptor					
	Spotted Sandpiper	**Hooded Merganser**	**Bald Eagle**	**Osprey**	**Mink**	**River Otter**
Metals						
Antimony					X	
Arsenic	X					
Cadmium	X					
Chromium	X					
Copper	X	X			X	
Lead	X	X	X	X	X	
Mercury	X	X	X	X	X	
Selenium	X				X	
Thallium	X					
Zinc	X					
Butyltins						
Butyltin ion	X					
Tributyltin ion	X					
PAHs						
Benzo(a)pyrene	X	X		X		
Total PAHs	X				X	X
Phthalates						
BEHP	X	X	X	X		
Dibutyl phthalate	X					
PCBs						
PCB TEQ	X	X	X	X	X	X
Total PCBs	X	X	X	X	X	X
Dioxins						
Dioxin TEQ	X	X	X	X	X	X
Pesticides						
Aldrin	X					
Sum DDD	X					
Sum DDE	X	X	X	X		
Sum DDT	X	X	X	X		
Total DDTs	X	X			X	X

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

Table 5-6. Exposure Parameters Used for Wildlife Dietary Risk Calculations

Receptor	BW (kg)	FIR (kg ww/ day)	SI (%)[a]	SIR (kg dw/day)[b]	SUF
Spotted sandpiper	0.047	0.0548	18	0.00148	1.0[c]
Hooded merganser	0.54	0.200	2	0.00110	1.0[c]
Bald eagle	4.5	0.540	2	0.00281	1.0[c]
Osprey	1.88	0.395	2	0.00205	1.0[c]
Mink	0.974	0.156	9.4	0.00381	1.0
River otter	7.7	0.759	2	0.00465	1.0

Note: See Sections 5.2.2.2 through 5.2.2.7 for references and details on selected dietary parameters.

[a] Percent of incidental sediment ingestion of the dry diet.

[b] SIR = FIR x SI.

[c] An SUF of 0.5 was also used to estimate risks for bird receptors in the uncertainty section (Section 5.4.2).

BW – body weight

FIR – food ingestion rate

SI – sediment ingestion

SIR – sediment ingestion rate

SUF – site use factor

Table 5-7. Dietary Fractions of Prey Items Used in Wildlife Exposure Calculations

Receptor Species	Prey Item	Fraction of Prey Item in Diet
Spotted sandpiper	clam (field collected)	1.0 (Scenario 1)
	worm (laboratory bioaccumulation tissue)[a]	1.0 (Scenario 2)
Hooded merganser	clam (field-collected)	0.25
	crayfish	0.05
	sculpin	0.65
	peamouth[b]	0.05
Bald eagle	largescale sucker[c]	0.45
	carp	0.45
	peamouth[b]	0.05
	northern pikeminnow[d]	0.05
Osprey	largescale sucker[c]	0.83
	northern pikeminnow	0.07
	carp	0.06
	smallmouth bass	0.02
	black crappie[c]	0.01
	brown bullhead	0.01
Mink	crayfish	0.15
	largescale sucker[c]	0.15
	carp	0.15
	sculpin	0.15
	smallmouth bass	0.15
	juvenile chinook salmon	0.05
	peamouth[b]	0.05
	northern pikeminnow[d]	0.05
	brown bullhead	0.05
	black crappie[c]	0.05
River otter	crayfish	0.10
	clam (field-collected)	0.02
	carp	0.26
	largescale sucker[c]	0.26
	sculpin	0.26
	smallmouth bass	0.05
	black crappie[c]	0.05

[a] Crayfish concentration for thallium was used as a surrogate species for lab worms.

[b] Sculpin concentrations of PAHs, TEQ, and phthalates were used as surrogate species for peamouth.

[c] Carp concentrations of TEQ were used as a surrogate species for largescale sucker.

[d] Smallmouth bass concentrations of PAHs, TEQ, and phthalates were used as a surrogate species for northern pikeminnow.

[e] Sculpin concentrations of PAHs and phthalates were used as surrogate species for black crappie.

Table 5-8. Detail of Shorebird Beach Areas

Beach Area	Habitat Quality	Surface Sediment Samples	Field Clam Tissue Samples	Laboratory Worm Tissue Samples	Area (m^2)
B1[a]	high	B001, B003, B005	none	none	188,665
B2	high	B002	FC001	LW001	15,351
B3	low	B004	FC002	LW002	20,344
B4	high	B006	none	none	34,273
B5	high	B007, 03B031	FC003	LW003	53,108
B6	high	03B030, B008	FC004, FC005	LW004, LW005	40,573
B7	high	03B033	none	none	20,221
B8	high	04B024	none	none	16,635
B9	low	B010	FC008	LW008	6,004
B10	high	B011, B009	FC009	LW009	12,251
B11	high	B012	FC011	LW011	15,664
B12	low	04B023	FC012	FC012	14,828
B13	high	05B018	FC013	LW013	30,041
B14	low	B015	none	none	12,962
B15	high	06B022	FC016, 06R002	LW016	14,743
B16	high	B050, 07B024	FC017	LW017	34,141
B17	high	B018, 07B022	FC018, FC020, 07R003, 07R006	LW018, LW020	75,952
B18	low	B017	FC019	LW019	15,254
B19[a]	high	B019	FC021	LW021	15,295
B20[b]	high	B021, B022-1	FC024	LW024	46,669
B21	high	07B023	FC022	LW022	26,483
B22	high	B024	FC028	LW028	16,324
B23	low	B023	FC027-1	LW027-1, LW027-2	31,447
B24	high	B020, 09B024, 09B028, 08B032	FC026	LW026, LW029	52,756
B25	high	09B026	FC031	LW031	24,270
B26	low	B025-1	FC030	LW030	4,656
B27	high	B026	FC032	LW032	14,285
B28	high	09B027	FC033	LW033	23,919

[a] Sandpipers were observed at these shorebird beach areas during the shorebird reconnaissance survey conducted in June 2004.

[b] High quality shorebird habitat (i.e., stretches of sandy areas with some upland vegetation) at beach area B20 is divided into two segments, with rip-rap in-between the two high quality habitat segments.

Table 5-9. Round 2 COPCs for Which Benthic Tissue Concentrations were Modeled Using a Site-Specific Regression for Estimating Shorebird Dietary Exposures

Round 2 COPC	Clam Tissue Data		Worm Tissue Data	
	Sediment to Tissue Regression[a]	**r^2**	**Sediment to Tissue Regression[a]**	**r^2**
Metals				
Arsenic	NA	NA	NA	NA
Cadmium	y = 0.0674x + 0.0718	0.466	NA	NA
Chromium	NA	NA	NA	NA
Copper	NA	NA	NA	NA
Lead	y = 0.0025x + 0.0265	0.624	y = 0.0076x+0.156	0.666
Mercury	NA	NA	NA	NA
Selenium	NA	NA	NA	NA
Zinc	NA	NA	y = 0.0103x+24.52	0.162
PAHs				
Benzo(a)pyrene	$y = (0.176)*x^{0.514}$	0.438	$y = (0.587)* x^{0.409}$	0.176
Total PAHs	$y = (2.48)*x^{0.438}$	0.474	NA[b]	NA[b]
Phthalates				
BEHP	NA[b]	NA[b]	NA[b]	NA[b]
Dibutyl phthalate	NA[b]	NA[b]	NA[b]	NA[b]

[a] In the sediment-to-tissue regression equation, x = sediment concentration (dw values for metals, and OC-normalized values for organics) and y = tissue concentration (dw values for metals, and lipid-normalized values for organics).

[b] No regression was developed because no relationship was found between tissue and sediment in the dataset.

BEHP – bis(2-ethylhexyl) phthalate

COPC – chemical of potential concern

NA – not applicable

PAH – polycyclic aromatic hydrocarbon

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Table 5-10. Prey Tissue EPCs Used to Estimate Dietary Dose Exposure for Hooded Merganser, Osprey, Bald Eagle, Mink, and River Otter

Round 2 COPC	Unit	Prey Tissue EPCs										
		Black Crappie	Brown Bullhead	Carp	Chinook	Clam	Crayfish	Largescale Sucker	Northern Pikeminnow	Peamouth	Sculpin	Smallmouth Bass
Antimony	mg/kg	0.0005[a]	0.0005[a]	0.0016	0.0011	0.0022	0.0091	0.0029	0.00089	0.0032	0.0017	0.0012
Copper	mg/kg	0.977	0.76	1.29	1.3	10	15	1.0	0.76	1.7	1.3	0.78
Lead	mg/kg	0.038	0.035	0.18	0.019	0.11	0.34	0.16	0.013	37	0.20	0.26
Mercury	mg/kg	0.046	0.046	0.045	0.016	0.010	0.030	0.081	0.40	0.051	0.047	0.092
Selenium	mg/kg	0.3[a]	0.30	0.35	0.18	0.13	0.15	0.3[a]	0.31	0.45	0.18	0.23
Benzo(a)pyrene	µg/kg	16[b]	33[a]	16	27	240	17	16	16[c]	16[b]	16	16
Total PAHs	µg/kg	380[b]	620	930	120	891	410	770	680[c]	380[b]	380	680
BEHP	µg/kg	13,000[b]	4,900	49	350	76	72	5,500	73,000[c]	13,000[b]	13,000	73,000
PCB TEQ (bird)	pg/g	51	9.4	28	8.3	25	23	28[d]	42[c]	97[b]	97	42
PCB TEQ (mammal)	pg/g	5.85	11.1	35.9	3.44	3.6	2.95	35.9[d]	20[c]	24.2[b]	24.2	20
Total PCBs	µg/kg	333	1,460	6,470	152	576	170	1,540	1,350	279	1,300	1,960
Dioxin TEQ (bird)	pg/g	2.7	2.89	14	8.6	9.7	115	14[d]	12	202[b]	202	12
Dioxin TEQ (mammal)	pg/g	1.33	2.06	7.53	3.1	2.7	25	7.53[d]	5.5[c]	16.8[b]	16.8	5.5
Sum DDE	µg/kg	82	61	192	67	35	15	160	412	176	320	160
Sum DDT	µg/kg	22	44	83	26	49	6.8	270	190	13	240	66
Total DDTs	µg/kg	112	119	310	190	130	27	624	593	214	1,400	262

[a] Maximum value was used to represent the tissue EPC because no UCL was recommended from ProUCL because of insufficient number of unique values.

[b] Sculpin concentrations of dioxin TEQ, PCB TEQ, PAHs, and/or phthalates were used as a surrogate to represent TEQ concentrations for peamouth and black crappie.

[c] Smallmouth bass concentrations of dioxin TEQ, PCB TEQ, PAHs, and phthalates were used as a surrogate to represent TEQ, PAH, and phthalate concentrations for northern pikeminnow.

[d] Carp concentrations of dioxin TEQ and PCB TEQ were used as a surrogate to represent TEQ concentrations for largescale sucker.

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
EPC – exposure point concentration
PAH – polycyclic aromatic hydrocarbon
TEQ – toxic equivalent

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Table 5-11. Metals and TBT EPCs in Benthic Prey Tissues Used to Estimate Spotted Sandpiper Exposure

Area	Arsenic (mg/kg)		Cadmium (mg/kg)		Chromium (mg/kg)		Copper (mg/kg)		Lead (mg/kg)		Mercury (mg/kg)		Selenium (mg/kg)		Zinc (mg/kg)		TBT (µg/kg)	
	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm
Study Area	0.97	1.4	0.11	0.084	0.69	0.57	9.9	2.5	0.13	0.40	0.010	0.0062	0.13	0.29	37	27	15	11
B1	0.96[a]	1.4[a]	0.0833[b]	0.083[a]	0.7[a]	0.56[a]	10[a]	3.8[a]	0.0563[b]	0.246[b]	0.01[a]	0.006[a]	0.13[a]	0.29[a]	37[a]	25.3[b]	220[a]	540[a]
B2	1.06	1.25	0.12	0.0628	0.7	0.82	9.49	2.27	0.036	0.232	0.006	0.008	0.098	0.32	33.2	27.7	5.1	0.6
B3	1.25	0.807	0.218	0.229	0.79	0.67	11.1	2.58	0.071	0.517	0.008	0.0034	0.174	0.37	40.3	30.6	6.7	0.6
B4	0.96[a]	1.4[a]	0.0787[b]	0.083[a]	0.7[a]	0.56[a]	10[a]	3.8[a]	0.0393[b]	0.195[b]	0.01[a]	0.006[a]	0.13[a]	0.29[a]	37[a]	25.4[b]	220[a]	540[a]
B5	1.03	0.469	0.104	0.087	0.62	0.35	9.35	2.88	0.024	0.201	0.008	0.0034	0.114	0.35	30.5	26.1	4.7	2.6
B6	0.958	2.1	0.133	0.108	0.71	0.31	10.4	2.95	0.114	0.264	0.008	0.0053	0.123	0.31	48	25.5	63	20
B7	0.96[a]	1.4[a]	0.0758[b]	0.083[a]	0.7[a]	0.56[a]	10[a]	3.8[a]	0.0585[b]	0.253[b]	0.01[a]	0.006[a]	0.13[a]	0.29[a]	37[a]	25.2[b]	220[a]	540[a]
B8	0.96[a]	1.4[a]	0.0779[b]	0.083[a]	0.7[a]	0.56[a]	10[a]	3.8[a]	0.0565[b]	0.247[b]	0.01[a]	0.006[a]	0.13[a]	0.29[a]	37[a]	25.5[b]	220[a]	540[a]
B9	0.949	1.71	0.154	0.0494	0.43	0.55	10.5	1.83	0.261	0.847	0.01	0.005	0.087	0.085	29	23.3	7.2	3.1
B10	0.898	1.32	0.0953	0.0631	0.63	0.37	8.04	2.39	0.122	0.689	0.011	0.0047	0.098	0.27	33.5	24	8.7	2.3
B11	0.391[c]	1.83	0.0468[c]	0.0451	0.195[c]	0.3	3.38[c]	2.22	0.053[c]	0.186	0.0097[c]	0.0082	0.173[c]	0.31	10.8[c]	22.1	0.55[c]	0.46
B12	1.02	0.588	0.0761	0.0756	0.63	0.33	9.53	2.58	0.066	0.232	0.009	0.0042	0.088	0.31	38.1	30.7	4	0.22
B13	0.826	0.776	0.0685	0.0517	0.505	0.76	9.14	3.49	0.0685	0.407	0.01	0.004	0.17	0.09	31.8	28.4	8.9	2.7
B14	0.96[a]	1.4[a]	0.0784[b]	0.083[a]	0.7[a]	0.56[a]	10[a]	3.8[a]	0.0723[b]	0.295[b]	0.01[a]	0.006[a]	0.13[a]	0.29[a]	37[a]	25.1[b]	220[a]	540[a]
B15	0.965	1.68	0.0582	0.0528	0.59	0.42	11	2.07	0.115	0.274	0.012	0.0105	0.12	0.36	26.8	24.9	7.6	3
B16	1.05	1.13	0.0775	0.0499	0.73	0.75	11.6	2.2	0.072	0.329	0.013	0.006	0.1	0.09	36.9	30.3	2.1	0.95
B17	0.923	1.7	0.076	0.0421	0.63	0.56	9.5	2.31	0.316	0.287	0.011	0.0063	0.169	0.3	40.1	23.4	4.4	1.1
B18	0.913	3.04	0.0746	0.066	0.9	0.52	10.3	2.63	0.073	0.329	0.011	0.0064	0.14	0.3	28.4	25.2	17	7.5
B19	0.946	1.94	0.0714	0.045	0.65	0.63	9.41	2.08	0.06	0.303	0.01	0.004	0.088	0.1	30.9	27.8	3.7	1.45
B20	0.963	1.02	0.112	0.0575	0.67	0.63	9.26	2.16	0.146	0.487	0.007	0.003	0.146	0.11	46.9	27.3	3.1	1
B21	0.83	0.985	0.0979	0.0432	0.57	0.36	8.52	2.32	0.119	0.206	0.011	0.004	0.09	0.11	42.4	25.1	33	49
B22	0.924	1.59	0.0925	0.0488	0.61	0.59	9.24	2.17	0.113	0.845	0.011	0.01	0.137	0.11	40.1	26.3	1.35	0.9
B23	0.868	1.39	0.0622	0.0405	0.71	0.78	8.42	2.66	0.054	0.373	0.008	0.007	0.10	0.21	32.4	24.3	3.5	7.3

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Table 5-11. Metals and TBT EPCs in Benthic Prey Tissues Used to Estimate Spotted Sandpiper Exposure

Area	Arsenic (mg/kg)		Cadmium (mg/kg)		Chromium (mg/kg)		Copper (mg/kg)		Lead (mg/kg)		Mercury (mg/kg)		Selenium (mg/kg)		Zinc (mg/kg)		TBT (µg/kg)	
	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm
B24	0.917	0.893	0.144	0.0812	0.45	0.7	9.36	3.27	0.192	0.44	0.012[c]	0.004	0.21	0.09	32.5	29.8	11[c]	22
B25	1.02	1.35	0.0658	0.0386	0.94	0.69	9.51	1.91	0.076	0.179	0.012	0.005	0.088	0.11	38.3	24	0.9[c]	0.75
B26	0.902	0.842	0.0641	0.0538	0.92	0.45	9.03	2.43	0.097	0.284	0.009	0.006	0.11	0.19	39.6	26.7	1.25[c]	0.37
B27	0.392[c]	0.969	0.0481[c]	0.0364	0.24[c]	0.37	3.6[c]	1.94	0.054[c]	0.279	0.0098[c]	0.004	0.081[c]	0.08	15.6[c]	31.1	0.55	1.25
B28	0.546[c]	1.81	0.0597[c]	0.0502	0.49[c]	0.61	4.66[c]	1.89	0.051[c]	0.31	0.012[c]	0.0091	0.247[c]	0.26	15.1[c]	23.1	0.6	0.49

[a] Tissue EPC represented using a UCL of the clam or worm tissue.

[b] Tissue EPC estimated from beach sediment concentration using a site-specific regression equation.

[c] Tissue EPC represented by laboratory bioaccumulation clam tissue; no field clam tissue data available.

EPC – exposure point concentration

TBT – tributyltin

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Table 5-12. PCBs, Dioxins, and SVOC EPCs in Benthic Prey Tissues Used to Estimate Spotted Sandpiper Exposure

Area	Benzo(a)pyrene (µg/kg)		Total PAHs (µg/kg)		BEHP (µg/kg)		Dibutyl phthalate (µg/kg)		PCB TEQ (pg/g)		Total PCBs (µg/kg)		Dioxin TEQ (pg/g)	
	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm
Study Area	200	110	699	12,000	82	130	570	260	26	371	622	2,790	10.5	362
B1	0.0107[a]	0.0305[a]	0.336[a]	16,000[b]	76[b]	140[b]	500[b]	230[b]	25[b]	310[b]	10.6[c]	10.3[c]	0.153[c]	0.375[c]
B2	4.7	9.0	301	329	27	140	8.0	8.0	22.3	76.6	157	402	1.66	4.19
B3	7.7	9.8	349	385	27	140	8.0	8.0	47.7	476	327	2970	2.04	9.02
B4	0.0006[a]	0.0031[a]	0.0408[a]	16,000[b]	76[b]	140[b]	500[b]	230[b]	25[b]	310[b]	19.7[c]	35.7[c]	0.155[c]	0.395[c]
B5	13	24	551	517	27	130	8.0	8.0	4.8	3.81	70.4	48.9	1.09	3.7
B6	16	37	491	992	27	150	8.0	8.0	28.7	21.3	306	302	1.95	4.77
B7	0.00121[a]	0.0054[a]	0.0839[a]	16,000[b]	76[b]	140[b]	500[b]	230[b]	25[b]	310[b]	11.9[c]	13.8[c]	9.7[b]	300[b]
B8	0.0107[a]	0.0305[a]	0.357[a]	16,000[b]	76[b]	140[b]	500[b]	230[b]	25[b]	310[b]	11.8[c]	13.7[c]	9.7[b]	300[b]
B9	9.8	79	300	1,840	27	27	8.0	8.0	9.39	15.2	99.5	147	8.86	39.2
B10	7.7	24	276	1,310	55	130	16.5	8.0	1.43[d]	7.44	21.2[d]	131	1.79	12
B11	1.0[d]	38	50.1[d]	1,210	70	110	21	8.0	9.09	9.11	127	108	3.35	3.93
B12	39	970	1,010	23,300	27	180	8.0	8.0	6.3	26.5	89.9	331	2.94	26.4
B13	4.6	28	165	1,380	55	27	16	135	7.48	10.1	153	282	1.56	4.67
B14	0.00751[a]	0.023[a]	0.267[a]	16,000[b]	76[b]	140[b]	500[b]	230[b]	0.952[c]	1.00[c]	10.5[c]	9.9[c]	0.163[c]	0.489[c]
B15	16.5	18	333	551	60	120	250	8.0	11.8	12.6	2,660	3,910	1.55	6.69
B16	60	140	1180	5,050	27	27	8.0	140	11.5	335	111	1,890	21.8	932
B17	19	100	574	5,080	170	200	240	470	9.09	19.6	176	208	32.1	124
B18	2.4	6.6	142	315	55	160	16	8.0	4.87	5.16	97.1	82.1	2.19	7.83
B19	2.8	12	136	675	27	27	8.0	115	8.17	18.9	90.1	166	0.68[d]	6.81
B20	1.9	34	143	1,000	27	27	8.0	115	10.7	20.4	183	488	2.5	5.1
B21	2.3	16	98.1	458	120	27	1,300	140	8.14	17	241	449	1.65[d]	5.89
B22	2.6	13	147	1,140	27	100	8.0	160	71.8	732	421	4,310	2.62	12.5
B23	1.4	3.5	34.8	174	27	55	8.0	140	3.46	4.38	85.2	117	1.02	2.61
B24	3.9	9.7	248	503	140	69	16	125	8.97	17	386	730	2.2	6.26
B25	1.8	2.2	49.7	83.6	27	27	8.0	135	4.13	3.95	61.3	67.6	1.18	1.88
B26	1.9	17	111	1,100	28	55	8.5	92	12.1	6.93	161	171	1.54	2.09

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Table 5-12. PCBs, Dioxins, and SVOC EPCs in Benthic Prey Tissues Used to Estimate Spotted Sandpiper Exposure

Area	Benzo(a)pyrene (µg/kg)		Total PAHs (µg/kg)		BEHP (µg/kg)		Dibutyl phthalate (µg/kg)		PCB TEQ (pg/g)		Total PCBs (µg/kg)		Dioxin TEQ (pg/g)	
	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm
B27	0.48[d]	4.4	36.3[d]	321	110[d]	27	8.0[d]	175	17.8	41.5	460	1,450	3.46	5.15
B28	1.8	3.7	59.5	147	140[d]	120	80[d]	46	4.23	4.73	50.1	73.7	0.75	1.91

[a] Tissue EPC estimated from beach sediment concentration using a site-specific regression equation.

[b] Tissue EPC represented using a 95 UCL of the clam or worm tissue.

[c] Tissue EPC estimated from beach sediment concentration using a FWM.

[d] Tissue EPC represented by laboratory bioaccumulation clam tissue; no field clam tissue data available.

EPC – exposure point concentration

PCB – polychlorinated biphenyl

SVOC – semivolatile organic compound

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Table 5-13. Pesticide EPCs in Benthic Prey Tissues Used to Estimate Spotted Sandpiper Exposure

Area	Aldrin (µg/kg)		Sum DDD (µg/kg)		Sum DDE (µg/kg)		Sum DDT (µg/kg)		Total DDTs (µg/kg)	
	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm	Clam	Worm
Study Area	1.1	14	130	611	37.7	66	55	20.7	139	725
B1	0.062[a]	0.069[a]	3.5[a]	3.7[a]	1.9[a]	2.0[a]	6.0[a]	17.6[a]	10.0[a]	19.5[a]
B2	0.171	0.335	9.49	15.5	12.4	25.3	1.61	0.186	23.5	41
B3	0.223	0.236	11.6	14.2	14.6	18	2.23	0.541	28.4	32.8
B4	0.061[a]	0.066[a]	1.1[a]	0.9[a]	1.8[a]	1.6[a]	0.61[a]	0.89[a]	2.1[a]	2.0[a]
B5	0.144	0.073	10.5	11.2	10.5	11.7	1.74	1.43	22.8	24.4
B6	0.187	0.431	10.7	23.1	12.3	21	3.4	0.635	26.4	44.5
B7	0.099[a]	0.171[a]	1.1[a]	0.95[a]	1.8[a]	1.8[a]	0.55[a]	0.69[a]	2.0[a]	1.8[a]
B8	0.099[a]	0.171[a]	2.4[a]	2.4[a]	2.3[a]	3.3[a]	0.59[a]	0.83[a]	4.3[a]	6.8[a]
B9	0.169	0.427	6.11	11.1	8.57	12.8	2.09	0.126	16.8	24.1
B10	0.0203[b]	0.634	0.974[b]	24.3	0.737[b]	27.4	0.109[b]	0.287	1.82[b]	52
B11	0.224	0.429	16.2	21.4	14.5	18.3	3.8	0.326	34.5	40.1
B12	0.217	0.0875	17.5	78.7	15.3	46	4.13	2.05	37	127
B13	0.383	0.255	11.3	27.2	15.4	16.7	1.58	2.83	28.3	46.7
B14	0.060[a]	0.064[a]	1.2[a]	0.96[a]	2.0[a]	2.4[a]	0.83[a]	1.6[a]	2.6[a]	3.2[a]
B15	0.5	0.627	6.42	20.9	9.45	28.2	4.15	0.501	19	49.6
B16	0.36	0.34	196	1,260	63.2	188	44.4	34.2	304	1490
B17	0.55	0.555	242	896	107	83.9	114	40	463	1020
B18	0.194	0.582	4.45	11.4	8.54	20.4	1.29	0.145	14.3	32
B19	0.243	0.467	14.5	29.6	14.7	25.8	3.83	0.407	33.1	55.8
B20	0.324	0.54	10.2	21.6	14.6	31.7	2.05	0.269	26.8	53.6
B21	0.29	0.451	7.83	17.6	13.4	23	2.43	0.113	23.6	40.7
B22	5.07	37	29.7	131	66.8	153	1.63	0.0973	98.1	284
B23	0.135	0.448	2.72	8.81	6.27	14.2	0.811	0.153	9.8	23.1
B24	0.29	0.463	6.76	10.7	10.3	12.2	1.81	0.271	18.9	23.1
B25	0.172	0.424	2.98	5.98	7.99	14.1	1.26	0.132	12.2	20.2
B26	0.37	0.378	4.81	12.8	12.8	15.5	1.22	1.63	18.9	29.9
B27	0.353	0.51	5.84	12.1	9.47	11.6	2.26	0.231	17.6	23.9
B28	0.126	0.407	2.15	4.78	4.89	9.66	0.778	0.0562	7.82	14.5

[a] Tissue EPC estimated from beach sediment concentration using a FWM.

[b] Tissue EPC represented by laboratory bioaccumulation clam tissue; no field clam tissue data available.

EPC – exposure point concentration

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Table 5-14. EPCs in Surface Sediment (Based on < 20-ft Depth) Used to Estimate Dietary Dose Exposure for All Wildlife Receptors, Except Spotted Sandpiper

Round 2 COPC	Unit	UCL
Arsenic	mg/kg	5.9
Cadmium	mg/kg	0.90
Chromium	mg/kg	42
Copper	mg/kg	72
Lead	mg/kg	67
Mercury	mg/kg	0.13
Selenium	mg/kg	2.5
Tributyltin ion	µg/kg	340
Benzo(a)pyrene	µg/kg	4,100
Total PAHs	µg/kg	66,000
BEHP	µg/kg	5,500
PCB TEQ (bird)	pg/g	401
PCB TEQ (mammal)	pg/g	32.4
Total PCBs	µg/kg	640
Dioxin TEQ (bird)	pg/g	3,360
Dioxin TEQ (mammal)	pg/g	1,580
Sum DDE	µg/kg	61
Sum DDT	µg/kg	520
Total DDTs	µg/kg	700

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
UCL – upper confidence limit

Table 5-15. EPCs in Surface Sediment for Metals and TBT Round 2 COPCs Used to Estimate Dietary Dose Spotted Sandpiper Exposures

Area	Arsenic (mg/kg)	Cadmium (mg/kg)	Chromium (mg/kg)	Copper (mg/kg)	Lead (mg/kg)	Mercury (mg/kg)	Selenium (mg/kg)	Zinc (mg/kg)	TBT (µg/kg)
All beaches	3.1	0.74	26	57	52	0.088	0.090	260	NA
B1	3.26	0.17	23.9	19.8	11.9	0.042	0.06	78.1	NA
B2	2.4	0.192	46.1	18.9	7.88	0.025	0.025	68.2	1.6[a]
B3	2.7	0.73	83.6	28.3	43.6	0.04	0.025	247	3.7[a]
B4	2.52	0.103	14	11.5	5.1	0.008	0.025	85.8	NA
B5	3.2	0.635	24	22.8	11.3	0.025	0.15	73	2.3[a]
B6	2.58	0.076	14.8	14.4	9.8	0.025	0.1	64.3	36[a]
B7	4	0.06	15	15.8	12.8	0.025	0.1	62	NA
B8	4.7	0.09	24	194	12	0.025	0.1	96	NA
B9	4.58	0.235	21.3	18.9	33.5	0.083	0.02	101	19[a]
B10	2.51	0.134	15.6	11.7	10.2	0.014	0.02	61	18[a]
B11	3.14	0.115	11.8	16.1	17.1	0.011	0.03	72.8	7.7[a]
B12	2.7	0.06	16	33.4	14	0.02	0.1	116	7.2[a]
B13	2.4	0.09	28	108	62	0.025	0.1	81	22[a]
B14	2.95	0.098	11.3	16.3	18.3	0.412	0.02	52.7	NA
B15	2.6	0.23	22	42.7	30	0.025	0.1	138	17[a]
B16	3.06	0.111	39.5	20.6	22.2	0.022	0.1	80	8.4[a]
B17	5.43	0.186	63.5	108	44.7	0.024	0.1	132	9.7[a]
B18	2.72	0.136	11	40.3	14.3	0.0035	0.08	79.4	1,600[a]
B19	3.82	0.11	13.3	20.9	7.88	0.017	0.025	76.9	9[a]
B20	3.62	4.21	24.3	53.5	242	0.174	0.07	1,230	10[a]
B21	0.7	0.09	19	69.6	15	0.025	0.1	97	170[a]
B22	8	0.535	26.8	60.5	73.1	0.101	0.08	272	10[a]
B23	1.67	0.098	14.8	25.1	14	0.015	0.025	76.6	30.5
B24	2.2	0.116	22.3	23.9	12	0.046	0.1	78	150[a]
B25	2.4	0.05	13.2	18.3	11	0.025	0.1	121	3.6[a]
B26	2.51	0.23	14.7	18.5	21.4	0.0343	0.025	129	1.8[a]
B27	2.84	0.24	30.4	27.1	30.8	0.047	0.02	284	10[a]
B28	1.3	0.065	21.5	22.5	8.95	0.02	0.1	69	1.4[a]

[a] Sediment data from beach transects were not available because Round 2 COPC was not analyzed; sediment concentrations were represented using the results of the co-located sediment data that was collected with benthic clam and worm tissue for bioaccumulation testing.

COPC – chemical of potential concern

NA – not available

TBT – tributyltin

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Table 5-16. EPCs in Surface Sediment for Organic Round 2 COPCs Used to Estimate Dietary Dose Spotted Sandpiper Exposures

Area	Total PCBs (µg/kg)	Dioxin TEQ (pg/g)	Benzo(a)-pyrene (µg/kg)	Total PAHs (µg/kg)	PCB TEQ (pg/g)	Aldrin (µg/kg)	Sum DDE (µg/kg)	Sum DDT (µg/kg)	Total DDTs (µg/kg)	BEHP (µg/kg)	Dibutyl-phthalate (µg/kg)
All beaches	540	31.7	330	2,210	250	0.84	52	110	200	160	200
B1	1.2	0.139	360	3,120	NA	0.0167	0.304	11.8	17.8	7.0	3.9
B2	194	0.434	6.6	53.3	57.6	0.0152	0.0219	0.474	0.551	2.8	3.5
B3	1,770	2.72	82	694	549	0.01555	0.0225	6.69	8.32	5.0	14
B4	20.8	0.185	1.3	25.3	NA	0.01455	0.021	0.332	0.406	3.2	1.55
B5	19.7	0.383	59	549	0.495	0.095	0.43	0.471	2.11	45	9.5
B6	7.95	0.482	7.9	170	0.444	0.28	0.195	0.259	0.883	61	9.5
B7	3.9	ND	5.2	131	NA	0.095	0.195	0.195	0.195	46	9.5
B8	3.85	ND	360	3,570	NA	0.095	1.2	0.29	5.18	65	9.5
B9	31.3	7.62	900	7,270	8.88[a]	0.857	0.267	1.02	1.85	260	6.6
B10	1.05	20	9.9	105	2.73[a]	0.0141	0.0204	1.23	1.3	2.4	5.2
B11	1.15	15.3	110	918	2.32[a]	0.0142	0.285	0.802	3.16	1.0	1.5
B12	15.1	2.92[a]	42	489	4.12[a]	0.095	0.45	0.945	4.3	46	10
B13	29.4	0.747[a]	86	887	4.02[a]	0.095	0.6	0.7	0.7	230	27
B14	0.95	0.407	180	1,840	0.153	0.0132	0.602	0.805	1.63	4.0	4.2
B15	14.5	1.24[a]	4.4	242	73.4[a]	0.1	0.255	0.25	0.255	9.5	35
B16	78.8	80.1[a]	53	565	48.9[a]	2.48	104	139	372	59	9.5
B17	163	248	16	145	13.1	0.932	135	307	490	27	690
B18	1.1	0.356	15	109	0.276	0.01475	0.104	0.304	0.505	7.0	3.6
B19	1.05	0.817	2	24.3	4.71[a]	0.01405	0.0203	2.9	3.54	7.0	4.6
B20	126	5.35	1,300	11,900	12.6	0.01625	2.43	1.48	6.02	2600	9.5
B21	12.5	1.73[a]	15	227	18[a]	0.095	21.2	61	118	69	27
B22	83	4.83	22	254	20.4	0.01585	5.05	0.812	13.5	39	8.5
B23	35.6	0.422	6.3	70.6	0.731	0.0137	0.422	0.528	1.23	12.5	3.5
B24	37.2	0.596	16	266	1.46	0.095	0.528	0.742	2.35	68	26
B25	3.9	0.52[a]	0.95	99.6	0.903[a]	0.095	0.195	0.195	0.195	53	9.5

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Table 5-16. EPCs in Surface Sediment for Organic Round 2 COPCs Used to Estimate Dietary Dose Spotted Sandpiper Exposures

Area	Total PCBs (µg/kg)	Dioxin TEQ (pg/g)	Benzo(a)-pyrene (µg/kg)	Total PAHs (µg/kg)	PCB TEQ (pg/g)	Aldrin (µg/kg)	Sum DDE (µg/kg)	Sum DDT (µg/kg)	Total DDTs (µg/kg)	BEHP (µg/kg)	Dibutyl-phthalate (µg/kg)
B26	119	1.84	4.4	58.2	4.99[a]	0.0141	0.201	0.345	0.745	3.5	4.95
B27	362	0.897	250	2,710	17.2	0.01905	2.2	13.8	19.3	170	10
B28	24.8	0.489[a]	9	172	2.11[a]	0.1	0.6	0.47	0.6	21	10

[a] Sediment data from beach transects were not available because COPC was not analyzed; sediment concentrations were represented using the results of the co-located sediment data that was collected with benthic clam and worm tissue for bioaccumulation testing.

BEHP – bis(2-ethylhexyl) phthalate

COPC – chemical of potential concern

NA – not available

PAH – polycyclic aromatic hydrocarbon

TEQ – toxic equivalent

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Table 5-17. Estimated Dietary Doses (IR_{dose}) of Round 2 COPCs for Hooded Merganser, Bald Eagle, Osprey, Mink, and River Otter

Round 2 COPC	Unit (bw/day)	IR_{dose}				
		Hooded Merganser	Bald Eagle	Osprey	Mink	River Otter
Antimony	mg/kg	NE	NE	NE	0.00729	NE
Copper	mg/kg	1.69	NE	NE	NE	NE
Lead	mg/kg	0.886	NE	NE	0.587	NE
Mercury	mg/kg	0.0140	0.00959	0.0213	0.0121	NE
Selenium	mg/kg	NE	NE	NE	0.0512	NE
Benzo(a)pyrene	µg/kg	35.0	NE	7.87	NE	NE
Total PAHs	µg/kg	NE	NE	NE	352	104
BEHP	µg/kg	3,390	819	2,384	NE	NE
PCB TEQ	ng/kg	28.7	4.11	6.59	3.50	2.64
Total PCBs	µg/kg	376	443	383	306	253
Dioxin TEQ	ng/kg	62.2	4.89	6.52	7.91	2.06
Sum DDE	µg/kg	83.9	22.6	37.4	NE	NE
Sum DDT	µg/kg	63.7	20.6	51.9	NE	NE
Total DDTs	µg/kg	355	55.7	124	75.6	62.6

BEHP – bis(2-ethylhexyl) phthalate
bw – body weight
COPC – chemical of potential concern
NE – not evaluated; not identified as a Round 2 COPC/receptor pair
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

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Table 5-18. Estimated Dietary Doses (IR_{dose}) of Metals and TBT Round 2 COPCs for Spotted Sandpiper

Beach Area	IR_{dose}																	
	Arsenic (mg/kg bw/day)		Cadmium (mg/kg bw/day)		Chromium (mg/kg bw/day)		Copper (mg/kg bw/day)		Lead (mg/kg bw/day)		Mercury (mg/kg bw/day)		Selenium (mg/kg bw/day)		Zinc (mg/kg bw/day)		TBT ion (µg/kg bw/day)	
	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey
All beaches	1.23	1.1.73	0.152	0.121	1.62	1.48	13.3	4.71	1.79	2.10	0.0144	0.0100	0.154	0.341	51.3	39.7	ND	ND
B1	1.22	1.74	0.103	0.102	1.57	1.40	12.3	5.06	0.440	0.662	0.0130	0.00833	0.154	0.341	45.7	32.0	ND	ND
B2	1.31	1.54	0.146	0.0794	2.27	2.41	11.7	3.24	0.290	0.519	0.00779	0.0101	0.115	0.374	40.9	34.5	6.01	0.751
B3	1.54	1.03	0.278	0.290	3.55	3.41	13.9	3.90	1.45	1.97	0.0106	0.00523	0.204	0.433	54.8	43.5	7.94	0.817
B4	1.20	1.71	0.0952	0.100	1.26	1.09	12.0	4.80	0.206	0.388	0.0119	0.00726	0.153	0.339	45.9	32.4	ND	ND
B5	1.30	0.648	0.141	0.122	1.48	1.16	11.6	4.08	0.383	0.590	0.0101	0.00476	0.138	0.413	37.9	32.8	5.56	3.11
B6	1.20	2.53	0.158	0.129	1.29	0.827	12.6	3.90	0.441	0.616	0.0101	0.00697	0.147	0.365	58.1	31.8	74.7	24.5
B7	1.25	1.76	0.0905	0.0988	1.29	1.13	12.2	4.93	0.471	0.698	0.0125	0.00779	0.155	0.342	45.2	31.3	ND	ND
B8	1.27	1.78	0.0938	0.997	1.57	1.41	17.8	10.5	0.443	0.666	0.0125	0.00779	0.155	0.342	46.2	32.8	ND	ND
B9	1.25	2.14	0.187	0.065	1.17	1.31	12.9	2.73	1.357	2.04	0.0143	0.00845	0.102	0.100	37.0	30.4	9.00	4.22
B10	1.13	1.62	0.115	0.0779	1.23	0.922	9.76	3.16	0.463	1.13	0.0133	0.00593	0.115	0.316	41.0	29.9	10.7	3.25
B11	0.555	2.24	0.0583	0.0563	0.598	0.721	4.45	3.10	0.599	0.755	0.0116	0.00992	0.203	0.363	14.9	28.1	0.884	0.779
B12	1.28	0.771	0.0907	0.0902	1.24	0.888	12.2	4.06	0.517	0.711	0.0111	0.00553	0.106	0.365	48.1	39.5	4.90	0.483
B13	1.04	0.982	0.0828	0.063	1.47	1.77	14.1	7.47	2.03	2.42	0.0125	0.00546	0.202	0.108	39.7	35.7	11.1	3.84
B14	1.21	1.73	0.0946	0.100	1.17	1.01	12.2	4.95	0.659	0.920	0.0246	0.0200	0.152	0.339	44.9	30.9	ND	ND
B15	1.21	2.04	0.0752	0.0689	1.38	1.18	14.2	3.76	1.08	1.26	0.0148	0.0130	0.143	0.424	35.6	33.4	9.41	4.04
B16	1.32	1.42	0.0940	0.0618	2.09	2.12	14.2	3.22	0.782	1.08	0.0159	0.00770	0.120	0.108	45.6	37.9	2.72	1.37
B17	1.25	2.16	0.0946	0.0550	2.73	2.65	14.5	6.09	1.77	1.74	0.0136	0.00811	0.200	0.353	51.0	31.5	5.44	1.59
B18	1.15	3.64	0.0914	0.0813	1.40	0.953	13.3	4.34	0.535	0.834	0.0130	0.00758	0.166	0.353	35.7	31.9	70.1	59.0
B19	1.22	2.39	0.0868	0.0560	1.18	1.15	11.6	3.09	0.318	0.601	0.0122	0.00521	0.104	0.118	38.5	34.9	4.60	1.98
B20	1.24	1.30	0.263	0.199	1.55	1.50	12.5	4.20	7.78	8.17	0.0136	0.00897	0.173	0.131	93.4	70.5	3.93	1.48
B21	0.991	1.17	0.117	0.0533	1.26	1.02	12.1	4.90	0.610	0.712	0.0136	0.00546	0.108	0.132	52.6	32.4	43.9	62.6
B22	1.33	2.11	0.125	0.0738	1.55	1.53	12.7	4.44	2.43	3.28	0.0160	0.0149	0.162	0.131	55.4	39.3	1.89	1.37

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Table 5-18. Estimated Dietary Doses (IR_{dose}) of Metals and TBT Round 2 COPCs for Spotted Sandpiper

Beach Area	IR_{dose}																	
	Arsenic (mg/kg bw/day)		Cadmium (mg/kg bw/day)		Chromium (mg/kg bw/day)		Copper (mg/kg bw/day)		Lead (mg/kg bw/day)		Mercury (mg/kg bw/day)		Selenium (mg/kg bw/day)		Zinc (mg/kg bw/day)		TBT ion (µg/kg bw/day)	
	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey
B23	1.07	1.68	0.0757	0.0504	1.29	1.38	10.6	3.89	0.503	0.875	0.00981	0.00865	0.117	0.246	40.2	30.8	5.03	9.47
B24	1.14	1.11	0.172	0.0985	1.23	1.52	11.7	4.57	0.601	0.891	0.0157	0.00612	0.248	0.108	40.4	37.2	17.6	30.4
B25	1.27	1.65	0.0784	0.0466	1.51	1.22	11.7	2.81	0.434	0.555	0.0148	0.00662	0.106	0.132	48.5	31.8	1.16	0.989
B26	1.13	1.06	0.0821	0.0701	1.54	0.987	11.1	3.42	0.786	1.00	0.0116	0.00808	0.129	0.223	50.3	35.2	1.52	0.489
B27	0.547	1.22	0.0637	0.0500	1.24	1.39	5.06	3.12	1.03	1.29	0.0129	0.00615	0.0952	0.0940	27.1	45.2	0.956	1.77
B28	0.678	2.15	0.0718	0.0607	1.25	1.39	6.15	2.91	0.341	0.643	0.0151	0.0113	0.292	0.307	19.8	29.1	0.745	0.616

bw – body weight
COPC – chemical of potential concern
ND – no sediment or tissue data available to estimate shorebird exposure
TBT – tributyltin

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Table 5-19. Estimated Dietary Doses (IR_{dose}) of Pesticide Round 2 COPCs for Spotted Sandpiper

Beach Area	IR_{dose}									
	Aldrin (µg/kg bw/day)		Sum DDD (µg/kg bw/day)		Sum DDE (µg/kg bw/day)		Sum DDT (µg/kg bw/day)		Total DDTs (µg/kg bw/day)	
	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey
All beaches	1.31	16.3	153	714	45.6	78.6	67.6	27.6	168	852
B1	0.0729	0.0807	4.23	4.45	2.21	2.34	7.36	20.9	12.2	23.3
B2	0.200	0.392	11.1	18.1	14.5	29.5	1.89	0.232	27.5	47.9
B3	0.261	0.276	13.6	16.6	17.0	21.0	2.81	0.842	33.4	38.6
B4	0.0716	0.0774	1.25	1.01	2.05	1.84	0.722	1.05	2.46	2.36
B5	0.171	0.0882	12.3	13.1	12.3	13.7	2.05	1.68	26.7	28.6
B6	0.227	0.512	12.5	27.0	14.4	24.5	3.98	0.750	30.9	52.0
B7	0.118	0.202	1.34	1.11	2.15	2.15	0.643	0.809	2.34	2.10
B8	0.118	0.202	2.89	2.90	2.73	3.90	0.698	0.974	5.14	8.11
B9	0.224	0.526	7.15	13.0	10.0	15.0	2.47	0.179	19.7	28.2
B10	0.0241	0.741	1.14	28.4	0.861	32.0	0.166	0.374	2.17	60.8
B11	0.262	0.501	19.0	25.1	16.9	21.4	4.46	0.406	40.4	46.9
B12	0.256	0.105	20.5	92.0	17.9	53.7	4.85	2.42	43.3	148
B13	0.450	0.301	13.2	31.8	18.0	19.5	1.87	3.33	33.1	54.6
B14	0.0708	0.0753	1.36	1.13	2.39	2.86	0.997	1.87	3.14	3.82
B15	0.587	0.735	7.50	24.4	11.0	32.9	4.85	0.593	22.2	57.9
B16	0.498	0.475	233	1475	77.1	223	56.2	44.3	367	1750
B17	0.672	0.677	284	1048	129	102	143	56.4	556	1210
B18	0.227	0.680	5.20	13.3	9.98	23.8	1.52	0.179	16.7	37.4
B19	0.284	0.546	17.0	34.6	17.2	30.1	4.56	0.566	38.8	65.3
B20	0.379	0.631	12.0	25.3	17.1	37.1	2.44	0.361	31.5	62.8
B21	0.342	0.530	10.3	21.7	16.3	27.5	4.75	2.05	31.3	51.2
B22	5.92	43.2	34.9	153	78.2	179	1.93	0.139	115	332
B23	0.158	0.524	3.18	10.3	7.33	16.6	0.964	0.195	11.5	27.0
B24	0.342	0.544	7.93	12.5	12.0	14.3	2.14	0.340	22.1	27.0
B25	0.204	0.498	3.49	6.99	9.34	16.5	1.48	0.160	14.3	23.6
B26	0.433	0.442	5.62	15.0	15.0	18.1	1.44	1.91	22.1	34.9
B27	0.413	0.596	6.92	14.2	11.1	13.6	3.07	0.703	21.2	28.5
B28	0.150	0.478	2.52	5.59	5.73	11.3	0.923	0.0804	9.15	17.0

bw – body weight
COPC – chemical of potential concern

Table 5-20. Estimated Dietary Doses (IR_{dose}) of PCBs, Dioxins, PAHs, and Phthalate Round 2 COPCs for Spotted Sandpiper

Beach Area	IR_{dose}													
	Benzo(a)pyrene (µg/kg bw/day)		Total PAHs (µg/kg bw/day)		BEHP (µg/kg bw/day)		Dibutyl phthalate (µg/kg bw/day)		PCB TEQ (pg/kg bw/day)		Total PCBs (µg/kg bw/day)		Dioxin TEQ (pg/kg bw/day)	
	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey
All beaches	244	139	885	14,100	101	157	671	309	38.2	440	742	3,270	13.2	423
B1	11.3	11.3	98.4	18,800	89.0	164	584	269	ND	ND	12.4	12.0	0.183	0.443
B2	5.70	10.7	353	386	31.0	164	9.45	9.45	27.9	91.3	189	476	1.95	4.91
B3	11.6	14.0	429	471	31.1	164	9.78	9.78	73.0	573	437	3520	2.47	10.6
B4	0.0415	0.0444	0.843	18,700	88.8	164	584	269	ND	ND	23.7	42.4	0.187	0.467
B5	17.0	29.9	661	621	32.4	153	9.64	9.64	5.62	4.46	82.8	57.7	1.28	4.33
B6	18.9	43.5	579	1,164	32.9	177	9.64	9.64	33.5	24.9	358	353	2.29	5.59
B7	0.165	0.170	4.21	18,700	90.2	165	584	269	ND	ND	14.0	16.2	ND	ND
B8	11.3	11.3	113	18,800	90.8	166	584	269	ND	ND	13.9	16.1	ND	ND
B9	39.7	121	579	2,380	39.1	39.1	9.55	9.55	11.2	18.0	117	173	10.6	46.0
B10	9.30	28.3	326	1,530	64.3	152	19.4	9.51	1.76	8.8	24.8	153	2.72	14.6
B11	4.62	47.8	87.3	1,440	81.8	128	24.6	9.39	10.7	10.7	148	126	4.39	5.07
B12	46.9	1130	1195	27,200	32.4	212	9.66	9.66	7.49	31.1	105	387	3.52	30.9
B13	8.07	35.4	221	1,640	71.5	38.2	19.5	158	8.86	11.9	180	330	1.85	5.48
B14	5.66	5.68	58.1	18,700	88.9	164	584	269	1.12	1.17	12.3	11.6	0.203	0.584
B15	19.4	21.2	396	651	70.4	140	293	10.4	16.1	17.0	3,110	4,570	1.85	7.85
B16	71.7	165	1,400	5,920	32.8	32.8	9.64	164	15.0	393	132	2,210	28.0	1,090
B17	22.7	117	675	5,940	199	234	302	571	11.0	23.3	211	248	45.3	153
B18	3.27	8.18	169	371	64.4	187	18.8	9.45	5.7	6.0	113	95.9	2.57	9.15
B19	3.33	14.1	160	789	31.2	31.2	9.49	134	9.7	22.2	105	194	0.820	7.98
B20	43.1	80.6	541	1,540	113	113	9.64	135	12.9	24.2	218	574	3.09	6.12
B21	3.16	19.2	122	542	142	33.1	1520	164	10.1	20.4	282	525	1.98	6.93
B22	3.73	15.9	180	1,340	32.2	118	9.61	187	84.5	855	494	5,040	3.21	14.7
B23	1.83	4.29	42.9	205	31.3	64.6	9.45	164	4.06	5.14	101	138	1.20	3.06
B24	5.06	11.8	298	596	166	82.7	19.5	147	10.5	19.9	452	854	2.59	7.33
B25	2.13	2.60	61.2	101	32.6	32.6	9.64	158	4.9	4.6	71.7	79.1	1.39	2.21

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Table 5-20. Estimated Dietary Doses (IR_{dose}) of PCBs, Dioxins, PAHs, and Phthalate Round 2 COPCs for Spotted Sandpiper

Beach Area	IR_{dose}													
	Benzo(a)pyrene (µg/kg bw/day)		Total PAHs (µg/kg bw/day)		BEHP (µg/kg bw/day)		Dibutyl phthalate (µg/kg bw/day)		PCB TEQ (pg/kg bw/day)		Total PCBs (µg/kg bw/day)		Dioxin TEQ (pg/kg bw/day)	
	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey	Clam Prey	Worm Prey
B26	2.36	20.0	131	1,290	32.8	64.3	10.1	108	14.3	8.2	192	203	1.86	2.50
B27	8.42	13.0	128	460	134	36.3	9.66	205	21.3	49.0	549	1,700	4.07	6.04
B28	2.38	4.60	74.9	177	164	141	93.7	54.0	5.01	5.59	59.3	86.8	0.891	2.25

BEHP – bis(2-ethylhexyl) phthalate
bw – body weight
COPC – chemical of potential concern
ND – no sediment or tissue data available to estimate shorebird exposure
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

Table 5-21. 2,3,7,8-TCDD Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (µg/kg bw/day)	LOAEL (µg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
2,3,7,8-TCDD	ring-necked pheasant	**0.04**	**0.14**	weekly IP injection for 10 weeks	body weight loss; reduced adult survival, egg production, and embryo survival	Nosek et al. (1992)
2,3,7,8-TCDD	chicken	0.1	1.0	oral intubation for 20 to 21 days	reduced survival (8 out of 10 died)	Schwetz et al. (1973)
2,3,7,8-TCDD	chicken	1.0		oral intubation for 20 to 21 days	no effect on growth	Schwetz et al. (1973)

IP – intraperitoneal
LOAEL – lowest-observed-adverse-effects level
NOAEL – no-observed-adverse-effects level
TCDD – tetrachlorodibenzo-*p*-dioxin

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Bold identifies the NOAEL and LOAEL selected as the TRVs.

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Table 5-22. PCB Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Aroclor 1254	chicken	0.054		diet for 20 weeks	no effect on fertility, hatchability, growth, or survival	Ahmed et al. (1978)
Aroclor 1248	American kestrel		0.35	diet for 5.5 months	reduced eggshell weight and thickness	Lowe and Stendell (1991)
Aroclor 1248	screech owl	0.49		diet for two generations	no effect on eggshell thickness, egg production, hatching success, or fledging success	McLane and Hughes (1980)
Aroclor 1242	chicken	**0.29**	**0.58**	diet for 6 weeks + 5 weeks untreated	reduced hatchability	Britton and Huston (1973)
Aroclor 1242	Japanese quail		0.60	diet for 45 days	eggshell thinning	Hill et al. (1975)
Aroclor 1248	chicken	0.061	0.61	diet for 8 weeks	reduced egg production and egg hatchability	Scott et al. (1975)
Aroclor 1232	chicken		1.2	diet for 9 weeks	reduced hatchability, embryo abnormality, and reduced survival	Cecil et al. (1974)
Aroclor 1254	ringed turtle dove		1.4	diet for two generations	reduced hatching success in second generation	Peakall et al. (1972); Peakall and Peakall (1973)
Aroclor 1254	ring-necked pheasant		1.6	gelatin capsule once per week for 16 weeks	reduced egg hatchability	Dahlgren et al. (1972)
Aroclor 1254	mourning dove		1.6	diet for 42 days	reduced courtship behavior, fewer successful pair bonds formed; delayed onset of nest initiation	Tori and Peterle (1983)
Aroclor 1254	mallard	2.5		diet for ~1 month	reproductive success	Custer and Heinz (1980)
Aroclor 1254	chicken	0.29	2.9	diet for 39 weeks	reduced hatchability	Platonow and Reinhart (1973)
Aroclor 1254	mallard	3.9		diet for 4 months	no effect on egg production or eggshell thinning	Risebrough and Anderson (1975)
Aroclor mixture	American kestrel		5 to 7	diet for 100 days	reduced egg laying in F2 generation; some effect on clutch size and fledgling success	Fernie et al. (2000; 2001)
Aroclor mixture	American kestrel		5 to 7	diet during reproductive period	altered courtship behavior	Fisher et al. (2001)

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Table 5-22. PCB Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Aroclor mixture	American kestrel		5 to 7	diet during reproductive period	increase in number of cracked eggs, embryo abnormalities, reduced offspring survival (48%), reduced offspring growth rate in F2 generation.	Fernie et al. (2003a; 2003b; 2003c)[a]
Aroclor 1254	American kestrel		9.5	diet for 62 to 69 days	decreased sperm count and sperm concentration	Bird et al. (1983)
Aroclor 1254	Japanese quail		12.0	diet for 8 days	avoidance response	Kreitzer and Heinz (1974)
Aroclor 1242	mallard		15	diet for 12 weeks	reduced hatchability, embryo survival, egg viability; embryo abnormalities	Haseltine and Prouty (1980)

[a] These studies have been added to the TRV review process (Attachment G5) since the submittal of the TRV technical memorandum to EPA as Appendix B of the Ecological PRE.

LOAEL – lowest-observed-adverse-effects level

NOAEL – no-observed-adverse-effects level

PCB – polychlorinated biphenyl

Bold identifies the NOAEL and LOAEL selected as the TRVs.

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Table 5-23. Arsenic Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Sodium arsenate	mallard	6.1		diet for 10 weeks	no effect on female growth or survival[a]	Camardese et al. (1990) as cited in Sample et al. (1996)
Copper acetoarsenite (As^{+3})	brown-headed cowbird	**2.3**	**6.8**	diet for 7 months[b]	reduced survival (80%)	USFWS (1969) as cited in Sample et al.(1996)
Sodium arsenate	mallard	40		diet for up to 128 days	no effect on body weight	Stanley et al. (1994)
Sodium arsenate	mallard	10	40	diet for up to 128 days	delayed egg laying; depressed egg weight and shell thinning; decrease offspring body weight and production[c]	Stanley et al. (1994)
Sodium arsenate	mallard (birth to 9 weeks)		42	diet for 9 weeks	altered resting and alertness behavior	Whitworth et al. (1991)
Sodium arsenite	mallard (young)	25	50	diet for 154 days	reduced survival (40%)[d]	USFWS (1964)

[a] Adverse effect on growth was observed; however, growth effect was recovered and does not warrant a LOAEL.
[b] Birds were exposed to multiple chemicals (both copper and arsenic).
[c] Reproductive effects may have been the result of insufficient food intake.
[d] There was 12 to 31% mortality in controls and 12% mortality at NOAEL.

bw – body weight
LOAEL – lowest-observed-adverse-effects level
NOAEL – no-observed-adverse-effects level
USFWS – US Fish and Wildlife Service
Bold identifies the NOAEL and LOAEL selected as the TRVs.

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Table 5-24. Cadmium Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Cadmium chloride	black duck		0.40	diet for 4 months prior to egg laying – 1 week after hatching	avoidance response in 7-day-old chicks[a]	Heinz and Haseltine (1983)
Cadmium chloride	mallard (young)	1.6		diet for 12 weeks	body weight, femur density	Cain et al. (1983)
Cadmium sulfate	chicken	**0.73**	**2.9**	diet for 48 weeks	reduced egg production, shell thickness	Leach et al. (1979)
Cadmium chloride	Japanese quail (chicks)		4.0	diet for 6 weeks	reduced male body weight	Richardson et al. (1974)[a]
Cadmium chloride	mallard	19		diet for 90 days	no effect on survival	White and Finley (1978b)
Cadmium chloride	mallard	20		diet for 30 to 90 days	no effect on body weight or adult survival	White and Finley (1978a)
Cadmium chloride	chicken (chicks)		24	diet for 21 days	reduced male body weight	Freeland and Cousins (1973)[a]
Cadmium chloride	mallard	1.5	20	diet for 30 to 90 days	reduced egg production[b]	White and Finley (1978a)
Cadmium chloride	chicken (chicks)		40	diet for 20 days	reduced male body weight	Pritzl et al. (1974)[a]
Cadmium chloride	mallard	16	47	diet for 42 day	reduced body weight	DiGiulio and Scanlon (1984)

[a] Avoidance response effect was no observed as a clear dose-response (no effect on avoidance response was observed at a higher dose level).

[b] Observed effects were not consistent with a clear dose-response relationship.

[c] These studies have been added to the TRV review process (Attachment G5) since the submittal of the TRV technical memorandum to EPA as Appendix B of the Ecological PRE.

LOAEL – lowest-observed-adverse-effects level

NOAEL – no-observed-adverse-effects level

Bold identifies the NOAEL and LOAEL selected as the TRVs.

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Table 5-25. Chromium Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Chromium picolinate	chicken	0.10		diet for 28 days	no effect on egg weight or shell thickness[a]	Lien et al. (2004)[b]
Chromium chloride	chicken	0.17		diet for 4 weeks	no effect on egg production or weight, interior egg quality	Jensen and Maurice (1980)
Chromium potassium sulfate (Cr^{3+})	black duck	**1.0**	**5.0**	diet for 10 months	reduced offspring duckling survival	Haseltine et al. unpub as cited in Sample (1996)[c]
Sodium chromate	chicken (chicks)	7.7		diet for 22 days	no effect on survival or adult male body weight	Romoser et al. (1961)
Chromium nitrate	herring gull (chicks)		22.8	single injection + 48 days untreated	depressed growth and altered behavior (balance, depth perception, thermo-regulation, and begging behavior)	Burger and Gochfeld (1995b)
Potassium chromate (Cr^{6+})	chicken (chicks)		105	diet for 2 weeks	reduced body weight, reduced adult survival	Chung et al. (1988)
Chromium sulfate (Cr^{6+})	chicken (chicks)		465	diet for 2 weeks	reduced body weight	Chung et al. (1988)
Chromium sulfate (Cr^{6+})	chicken (chicks)	465		diet for 2 weeks	no effect on survival	Chung et al. (1988)

[a] Effect of eggshell thinning on reproductive success (e.g., hatchability or offspring viability) was not reported.
[b] This study has been added to the TRV review process (Attachment G5) since the submittal of the TRV technical memorandum to EPA as Appendix B of the Ecological PRE.
[c] Original paper could not be located.
bw – body weight
LOAEL – lowest-observed-adverse-effects level
NOAEL – no-observed-adverse-effects level
Bold identifies the NOAEL and LOAEL selected as the TRVs.

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Table 5-26. Copper Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Effect Endpoint	Source
Copper sulfate	chicken (chicks)	2.1		diet for 17 days	no effect on body weight or survival	Dozier et al. (2003)[a]
Copper sulfate	chicken	11.2		diet for 90 days	no effect on damaged egg ratio or egg weight and survival	Balevi and Coskun (2004)[a]
Copper sulfate	chicken	15		diet for 28 days	no effect on egg weight or shell thickness	Lien et al. (2004)[a]
Copper sulfate	chicken (1 day old)	16	29	diet for 25 days	reduced growth	Smith (1969)
Copper sulfate	chicken (chicks)	21	41	diet for 4 weeks	reduced growth, gizzard erosion	Poupoulis and Jensen (1976)
Copper oxide	chicken (chicks)	**47**	**62**	diet for 10 weeks	reduced growth and survival	Mehring et al. (1960)
Copper chloride	chicken (chicks)		66	diet for up to 22 days	reduced body weight	Persia et al. (2004)[a]

[a] These studies have been added to the TRV review process (Attachment G5) since the submittal of the TRV technical memorandum to EPA as Appendix B of the Ecological PRE.

bw – body weight

LOAEL – lowest-observed-adverse-effects level

NOAEL – no-observed-adverse-effects level

Bold identifies the NOAEL and LOAEL selected as the TRVs.

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Table 5-27. Lead Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Lead acetate	Japanese quail (hatchings)		0.2	diet for 5 weeks	reduced egg production	Edens and Garlich (1983); Edens et al. (1976)
Lead acetate	Japanese quail, (6 weeks old)	0.2	2.0	diet for 5 weeks	reduced egg production	Edens and Garlich (1983)
Lead nitrate	mallard (first year)	2.5		diet for 12 weeks	no effect on survival, or occurrence of pathologic lesions	Finley et al. (1976)
Lead acetate	chicken	1.64	3.3	diet for 4 weeks	reduced egg production	Edens and Garlich (1983)
Metallic lead powder	American kestrel	5.82		diet for 5 to 7 months	no effect on survival, fertility, egg production, or eggshell thinning	Pattee (1984)
Lead acetate	ringed turtle dove	10.9		drinking water for 2 weeks prior to during reproduction	no effect on onset of egg laying, egg production, juvenile body weight, or egg fertility	Kendall and Scanlon (1981)
Lead acetate	Japanese quail	**2.0**	**20**	diet for 12 weeks	reduced egg hatchability	Edens et al. (1976)
Lead acetate	Japanese quail (chicks)	5.5	28	diet for 6 weeks	reduced body weight	Morgan et al. (1975)[a]
Lead acetate	ringed turtle dove	75		oral intubation for 7 days	no effect on growth	Kendall and Scanlon (1982)
Metallic lead powder	American kestrel (nestlings)	25	125	oral intubation for 10 days	reduced growth	Hoffman et al. (1985)
Lead acetate	herring gull (chicks)	50	100	single IP injection + 42 days untreated	altered thermoregulation behavior (increased time to reach shade)	Burger and Gochfeld (1995a)
Lead nitrate	herring gull (chicks)		100	single IP injection + 45 days untreated	altered behavior (righting response, balance, individual recognition, begging, thermoregulation, fish-swallowing speed, and depth perception)	Burger (1990)
Lead nitrate	herring gull (chicks)		100	single IP injection + 18 days untreated	altered behavior (response time, locomotor activity, required longer time to reach food)	Burger and Gochfeld (1993)

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Table 5-27. Lead Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Lead acetate	herring gull (chicks)		100	single IP injection + 21 days untreated	depressed growth (temporarily significant) and altered behavior (depressed begging and walking activity, increased error rate in pecking behavior while feeding)	Burger and Gochfeld (1988)
Lead nitrate	common tern (chicks)		200	single IP injection + 21 days untreated	depressed growth and reduced feeding efficiency (ability to manipulate fish)	Gochfeld and Berger (1988)
Lead nitrate	common tern (chicks)		200	two IP injections at day 2 and 7	altered behavior (locomotion, righting response, and depth perception)	Burger and Gochfeld (1988)
Lead nitrate	common tern (chicks)		200	single IP injection + 25 days untreated	depressed growth and altered behavior (locomotion, balance, righting response, feeding tasks, depth perception, and thermoregulation)	Burger and Gochfeld (1985)
Lead nitrate	common tern (chicks)		400	two IP injections at day 2 and 7	reduced survival (20%)	Burger and Gochfeld (1988)

[a] This study has been added to the TRV review process (Attachment G5) since the submittal of the TRV technical memorandum to EPA as Appendix B of the Ecological PRE.

bw – body weight

IP – intraperitoneal

LOAEL – lowest-observed-adverse-effects level

NOAEL – no-observed-adverse-effects level

Bold identifies the NOAEL and LOAEL selected as the TRVs.

BZTO104(e)030892

Table 5-28. Mercury Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Methylmercury dicyandiamide	mallard	**0.010**[a]	**0.050**	diet for multiple generations	reduced avoidance response in 2- and 8-day-old chicks	Heinz (1975)
Methylmercury	mallard	**0.010**[a]	**0.050**	diet for three generations	reduced egg and young production, eggshell thinning, duckling avoidance response and reduced response to maternal call behavior	Heinz (1979)
Methylmercury chloride	great egret (1 day old)		0.091	diet for 14 weeks	reduced growth	Spalding et al. (2000)
Methylmercury chloride	great egret (juvenile)		0.091	gelatin capsule for approx 6 weeks	reduced motivation to hunt prey, tendency to seek shade, altered maintenance behavior and activity	Bouton et al. (1999)
Methylmercury dicyandiamide	mallard	0.050	0.30	diet for 12 months through hatching	reduced hatching success, egg production, and reduced duckling survival to 1 week of age	Heinz (1974)
Methylmercury	mallard	0.050	0.30	diet for 2 years	reduced offspring survival, altered offspring avoidance response behavior	Heinz (1976)
Methylmercury chloride	Japanese quail		0.30	diet for 8 days treated + 6 days untreated	altered avoidance response behavior (depressed response to stimuli)	Kreitzer and Heinz (1974)
Methylmercury chloride	mallard	0.50		diet for > 60 days	no effect on eggshell thickness	Heinz (1980)
Methylmercury chloride	Japanese quail (chicks)		0.9	diet for 5 days	reduced hatchling survival (84%)	Hill and Soares (1987)
Methylmercury chloride	zebra finch	0.72	1.4	diet for 76 days	reduced survival	Scheuhammer (1988)
Methylmercury chloride	northern bobwhite (12 days old)	0.43	1.6	diet for 6 weeks	reduced survival	Spann et al. (1986)
Mercuric chloride	Japanese quail, (1 day old)	0.80	1.6	diet for 10 weeks	reduced eggshell thickness	Stoewsand et al. (1971)
Mercuric chloride	white corneaux pigeon	1.0	3.0	daily (inter-muscular) injections for up to 60 days	decreased learning response[b]	Leander et al. (1977)

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Table 5-28. Mercury Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Dimethyl mercury	American kestrel	5.24		diet for 3 months[c]	no effect on eggshell thickness	Peakall and Lincer (1972)
Methylmercury chloride	Japanese quail		6.4	gavage	reduced survival (LC50)	Hill and Soares (1987)
Methylmercury chloride	Japanese quail		8.8	injection	reduced survival (LC50)	Hill and Soares (1987)
Mercuric chloride	Japanese quail		11.1	gavage	reduced survival (LC50)	Hill and Soares (1987)
Mercuric chloride	Japanese quail		19.2	injection	reduced survival (LC50)	Hill and Soares (1987)
Mercuric chloride	Japanese quail (chicks)		62	diet for 5 days	reduced hatchling survival (88%)	Hill and Soares (1987)

[a] NOAEL extrapolated from LOAEL based on a UF of 5.

[b] Effect on learning response was inconsistently impaired at lowest dose (NOAEL) and consistently impaired at LOAEL.

[c] Dietary concentration (10 mg/kg) was assumed to be based on a wet weight (ww) basis and was converted into dry weight (dw) to calculate dietary dose assuming 80% moisture in diet of day-old chicks.

bw – body weight

LC50 – concentration that is lethal to 50% of an exposed population

LOAEL – lowest-observed-adverse-effects level

NOAEL – no-observed-adverse-effects level

Bold identifies the NOAEL and LOAEL selected as the TRVs.

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Table 5-29 Selenium Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Effect Endpoint	Source
Sodium selenite or sel-plex 50	broiler chicken (chicks)	0.025		diet for approximately 40 days	no effect on body weight	Choct et al. (2004)[a]
Seleno-methionine	mallard	**0.42**	**0.82**	diet for approximately 100 days	reduced offspring growth/survival	Heinz et al. (1989)
Sodium selenite	mallard	0.50	1.0	diet for 7 weeks during reproductive period	embryo abnormalities	Heinz et al. (1987)
Seleno-methionine	mallard	1.6		diet for approximately 100 days	no effect on body weight or survival	Heinz et al. (1989)
Sodium selenite	mallard	1.0	2.5	diet for 7 weeks during reproductive period	reduced adult growth	Heinz et al. (1987)
Seleno-methionine	screech owl	1.0	3.2	diet for approximately 3 months	reduced body weight, hatching success, 5-day offspring survival, clutch size, egg size, and mass[b]	Wiemeyer and Hoffman (1996)
Sodium selenite[c]	mallard	4.6		diet for 42 days	no effect on survival	Heinz et al. (1988)
Sodium selenite[c]	mallard	2.1	4.6	diet for 42 days	reduced body weight[b, b]	Heinz et al. (1988)
Sodium selenite	mallard	2.5	10	diet for 7 weeks during reproductive period	reduced adult survival (0%)	Heinz et al. (1987)
Sodium selenite[c]	mallard		10	diet for 42 days	reduced survival	Heinz et al. (1988)

[a] This study has been added to the TRV review process (Attachment G5) since the submittal of the TRV technical memorandum to EPA as Appendix B of the Ecological PRE.

[b] Food avoidance also observed and may have affected body weight gain (growth) endpoint.

[c] Selenomethionine was also administered to mallards in the diet; however, this chemical form of selenium was found to be less toxic than sodium selenite.

bw – body weight

LOAEL – lowest-observed-adverse-effects level

NOAEL – no-observed-adverse-effects level

Bold identifies the NOAEL and LOAEL selected as the TRVs.

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Table 5-30. Thallium Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Thallium sulfate	pheasant	**0.48**[a]	**24**	oral gavage for up to 14 days	reduced survival (LD50)	Hudson et al. (1984)
Thallium sulfate	golden eagle	17	34	gelatin capsule single dose + 21 days observed post-treatment	reduced survival (LD50)	Bean and Hudson (1976)[b]
Thallium sulfate	mallard		37	oral gavage for up to 14 days	reduced survival (LD50)	Hudson et al. (1984)

[a] NOAEL extrapolated from LOAEL based on a UF of 50.

[b] Study was uncontrolled study and a small sample size (n=3) was tested. Two of three of the birds died following the administration of one single dose and observation over 21 days after treatment.

bw – body weight

LD50 – dose that is lethal to 50% of an exposed population

LOAEL – lowest-observed-adverse-effects level

NOAEL – no-observed-adverse-effects level

Bold identifies the NOAEL and LOAEL selected as the TRVs.

Table 5-31. Zinc Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Zinc sulfate	chicken (chicks)	17		diet for 17 days	no effect on body weight or survival	Dozier et al. (2003)[c]
Zinc oxide, zinc sulfate, or zinc carbonate	chicken (chicks)	**82**[b]	**124**	diet for 5 weeks	reduced growth	Roberson and Schaible (1960)
Zinc carbonate	mallard (7 weeks old)		300	diet for 60 days	reduced survival; leg paralysis	Gasaway and Buss (1972)
Zinc chloride	chicken (chicks)		344	diet for 8 to 22 days	reduced body weight	Persia et al. (2004)[a]
Zinc acetate	chicken (chicks)	330	659	diet for 5 weeks	reduced survival and growth	Oh et al. (1979)
Zinc sulfate	chicken	133		diet for 44 weeks	no effect on egg hatchability	Stahl et al. (1990)

[a] This study has been added to the TRV review process (Attachment G5) since the submittal of the TRV technical memorandum to EPA as Appendix B of the Ecological PRE.

[b] NOAEL is based on background concentration in prepared food (30 ppm) plus exposure concentration added to food (120 ppm).

bw – body weight

LOAEL – lowest-observed-adverse-effects level

NOAEL – no-observed-adverse-effects level

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Bold identifies the NOAEL and LOAEL selected as the TRVs.

Table 5-32. TBT Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Effect Endpoint	Source
Tributyltin oxide	Japanese quail	**1.4**	**3.6**	diet for 6 weeks	reduced number of hatched eggs per pair[a]	Schlatterer et al. (1993)
Tributyltin oxide	Japanese quail	**1.4**	**3.6**	diet for 6 weeks	reduced embryo survival in shell, reduced hatchability	Coenen et al. (1992)
Tributyltin oxide	Japanese quail	22.5		diet for 6 weeks	no effect on body weight	Schlatterer et al. (1993)

[a] Effects on reproduction were more dramatic at a higher concentration (150 mg/kg in food) where egg production, egg fertility, hatchability, offspring body weight, and 14-day offspring survival were all significantly affected; however, effect on number of hatched eggs/ pair was significantly reduced in birds fed 60 mg/kg (LOAEL TRV).

bw – body weight

LOAEL – lowest-observed-adverse-effects level

NOAEL – no-observed-adverse-effects level

Bold identifies the NOAEL and LOAEL selected as the TRVs.

Table 5-33. Benzo(a)pyrene Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Benzo(a)pyrene	pigeon	**0.28[a]**	**1.4**	weekly intramuscular injection for 5 months	reduced fertility, ovarian appearance	Hough et al. (1993)
Benzo(a)pyrene	chicken	33		diet for 30 days	no effect on body weight gain	Rigdon and Neal (1963)

[a] A chronic NOAEL was extrapolated from the chronic LOAEL using a UF of 5.

bw – body weight

LOAEL – lowest-observed-adverse-effects level

NOAEL – no-observed-adverse-effects level

Bold identifies the NOAEL and LOAEL selected as the TRVs.

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Table 5-34. PAH Mixture Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Petroleum hydrocarbon mixture including PAHs	mallard	**40**		diet for 7 months[a]	no change in body weight following duration of study[b]	Patton and Dieter (1980)
Petroleum hydrocarbon mixture including PAHs	mallard	400		diet for 7 months[a]	no effect on survival	Patton and Dieter (1980)

[a] Food was dosed with a petroleum hydrocarbon mixture with paraffin wax in food. The aromatic mixture contained only some percentage of PAHs, not including benzo(a)pyrene.

[b] Body weight was significantly decreased at the NOAEL dose for three months, but at seven months, body weight reduction was recovered and body weight change was not significantly different than the control group.

LOAEL – lowest-observed-adverse-effects level

NOAEL – no-observed-adverse-effects level

PAH – polycyclic aromatic hydrocarbon

Bold identifies the NOAEL and LOAEL selected as the TRVs.

Table 5-35. Aldrin Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Aldrin	quail	**0.008**[a]	**0.04**	diet for 5 months	reduced survival (2.5%) in 127 days	DeWitt (1956)[b]
Aldrin	pheasant		0.7	diet for 16 to 20 weeks	0% survival in 46 days	DeWitt (1956)[b]
Aldrin	pheasant (chicks)	10	20	gelatin capsule for 7 weeks[c]	reduced growth	Hall et al. (1971)[b]
Aldrin	pheasant (chicks)		92	gelatin capsule for 7 weeks[d]	64% survival	Hall et al. (1971)[b]

[a] A chronic NOAEL was extrapolated from the chronic LOAEL using a UF of 5.

[b] These studies have been added to the TRV review process (Attachment G5) since the submittal of the TRV technical memorandum to EPA as Appendix B of the Ecological PRE.

[c] Birds were given one dose of a treated gelatin capsule per week. Effects were observed through 9 weeks following exposure period.

[d] Birds were given one dose of a treated gelatin capsule three times per week.

LOAEL – lowest-observed-adverse-effects level

NOAEL – no-observed-adverse-effects level

Bold identifies the NOAEL and LOAEL selected as the TRVs.

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Table 5-36. DDD Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Technical DDD	mallard	0.18[a]	0.90	diet for 2 years	reduced hatchling survival, production	Heath et al. (1969)
p,p′-DDD	ring-necked pheasant		34	diet for 8 days	reduced survival (LC50)	Gill et al. (1970)

[a] A chronic NOAEL was extrapolated from the chronic LOAEL using a UF of 5.

bw – body weight

LC50 – concentration that is lethal to 50% of an exposed population

LOAEL – lowest-observed-adverse-effects level

NOAEL – no-observed-adverse-effects level

Bold identifies the NOAEL and LOAEL selected as the TRVs.

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Table 5-37. DDE Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
"DDE"	barn owls	0.064[a]	0.32	diet for 2 years (two nestings)	increase in eggshell breakage/thickness; reduced nestling survival	Mendenhall et al. (1983)
"DDE"	barn owls[b]	0.32		diet for 2 years (two nestings)	no effect on adult survival	Mendenhall et al. (1983)
"DDE"	American kestrel		0.35	diet for 14 days	reduced shell thickness and egg permeability	Peakall et al. (1973)
p,p′-DDE	mallard		0.90	diet for 2 years	increased percentage of cracked eggs, reduced hatchling survival/production, and shell thickness	Heath et al. (1969)
p,p′-DDE	black duck		1.0	diet for 7 months	reduced shell thickness, egg weight, and hatchability; reduced duckling survival	Longcore and Samson (1973)
DDE	mallard		1.0	diet for 30 days	eggshell thinning	Kolaja (1977)
DDE	Black duck		1.0	diet during reproductive period	eggshell thinning and cracking, reduced embryo and duckling survival	Longcore et al. (1971)
DDE mixture	American kestrel[c]	0.12	1.2	diet for approximately 5 months	reduced eggshell thickness	Lincer (1975)
p,p′-DDE	American kestrel		1.0	diet for 1 year (2 clutches)	reduced eggshell thickness	Wiemeyer and Porter (1970); Porter and Wiemeyer 1972
"DDE"	ring dove		1.6	diet for 14 days	eggshell thinning, egg permeability	Peakall et al. (1973)
DDE	Japanese quail	2.99		diet for 8 days treated + 6 days untreated	no effect on avoidance response	Kreitzer and Heinz (1974)
"DDE"	white pekin duck		3.1	diet for 14 days	eggshell thinning, egg permeability	Peakall et al. (1973)
p,p′-DDE	white pekin duck		3.1	diet for 2 to 3 weeks	reduced eggshell thickness, breakage shell permeability, and egg yolk/shell gland weight	Pritchard et al. (1972)
p,p′-DDE	mallard		4.0	diet for 96 days	reduced eggshell thickness (first and second reproductive season)	Haegele and Hudson (1974)
p,p′ DDE	mallard		5.9	diet for 4 months	reduced eggshell thickness	Risebrough and Anderson (1975)

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Table 5-37. DDE Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
p,p′-DDE	ring dove		6.3	diet for 126 days	reduced egg production, shell thickness, survival of young, and clutch size	Haegele and Hudson (1973)
"DDE"	white Pekin duck		9.1	diet for 10 days	eggshell thinning	Peakall et al. (1975)
p,p′-DDE	Japanese quail	24		diet for 13 weeks	reduced body weight, egg production, eggshell thickness, egg weight, calcium content in eggshell;, and number and length of clutch	Davison et al. (1976)
p,p′-DDE	ring-necked pheasant		71	diet for 8 days	reduced survival (LC50)	Gill et al. (1970)

[a] A chronic NOAEL was extrapolated from the chronic LOAEL using a UF of 5.

[b] Three birds in the treated group died; however, the causes of two of the deaths were unknown and not related to DDE exposure

[c] Field-collected birds were used in this controlled laboratory experiment.

LC50 – concentration that is lethal to 50% of an exposed population

LOAEL – lowest-observed-adverse-effects level

NOAEL – no-observed-adverse-effects level

Bold identifies the NOAEL and LOAEL selected as the TRVs.

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Table 5-38. DDT Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
p,p′-DDT	quail	**0.030**[a]	**0.15**	diet for 26 weeks+12 days	reduced eggshell thickness, egg production	Stickel and Rhodes (1970)
p,p′-DDT	mallard	0.19	1.9	diet for 11 months	reduced eggshell weight and thickness	Davison and Sell (1974)
p,p′-DDT (technical grade)	bald eagle (field-collected)	[b]	0.44	diet for 5 months	reduced body weight (49% weight loss in adult male [n=1])	Chura and Stewart (1967)
DDT	mallard		1.0	diet for 30 days	eggshell thinning	Kolaja (1977)
p,p′-DDT	Japanese quail	0.6	1.5	diet for 26 weeks+12 days	reduced survival (67%)	Stickel and Rhodes (1970)
p,p′-DDT	mallard	1.9	27	diet for 11 months	reduced female survival (> 95%)	Davison and Sell (1974)
p,p′-DDT	mallard	0.90	2.3	diet for 2 years	reduced shell thickness, increased percentage of cracked eggs, reduced hatchling survival and production	Heath et al. (1969)
DDT-SO_4	mallard	5.0		diet for 30 days	eggshell thinning	Kolaja (1977)
p,p′-DDT	mallard	27		diet for 11 months	no effect on body weight	Davison and Sell (1974)
p,p′-DDT	ring-necked pheasant		36	diet for 8 days	reduced survival (LC50)	Gill et al. (1970)

[a] A chronic NOAEL was extrapolated from the chronic LOAEL using a UF of 5.

[b] Highly uncertain LOAEL because decrease in body weight does not appear dose-related, a very small sample size was used per dose (n=1 to 3 birds).

LC50 – concentration that is lethal to 50% of an exposed population

LOAEL – lowest-observed-adverse-effects level

NOAEL – no-observed-adverse-effects level

Bold identifies the NOAEL and LOAEL selected as the TRVs.

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Table 5-39. Total DDT Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
DDT mixture	double-crested cormorant	1.3		diet for 9 weeks	no effect on survival	Greichus and Hannon (1973)
tech DDT	mallard	**0.18**	**1.8**	diet for 11 months	reduced eggshell weight and thickness	Davison and Sell (1974)
tech DDT	mallard	1.8		diet for 11 months	reduced female survival (> 95%)	Davison and Sell (1974)
"DDT" (likely a mixture)	pheasant	2.3		diet for approximately 9 months	no effect on egg production	DeWitt (1956)
tech DDT	Japanese quail	3.0		diet for four generations	no effect on adult survival and body weight	Shellenberger (1978)
tech DDT	Bald eagle[a]	0.30	3.0	diet for 112 days	reduced survival (50%)	Stickel et al. (1966)
tech DDT	Japanese quail		3.0	diet for four generations	reduced fertility and hatchability	Shellenberger (1978)
DDT mixture	double-crested cormorant		3.1	diet for 9 weeks	reduced survival	Greichus and Hannon (1973)
"DDT" (likely a mixture)	pheasant		3.5	diet for approximately 8 months	reduced survival	DeWitt (1956)
"DDT" (likely a mixture)	ring-necked pheasant		4.5	diet for 8 weeks	reduced offspring survival	Genelly and Rudd (1956)
commercial DDT	white leghorn chicken	5.8		diet for 15 weeks	no effect on egg reproduction, hatchability, or strength	Scott et al. (1975); Scott et al. 1977
"DDT" (likely a mixture)	quail	6		diet for approximately 9 months	no effect on fertility	DeWitt (1956)
DDT mixture	double-crested cormorant	6.7		diet for 9 weeks	no effect on body weight	Greichus and Hannon (1973)
"DDT" (likely a mixture)	quail	7.2		diet for approximately 8 months	no effect on body weight	DeWitt (1956)
"DDT" (likely a mixture)	quail		8.9	diet for approximately 8 months	reduced survival	DeWitt (1956)

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Table 5-39. Total DDT Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
tech DDT	mallard	16		diet for 11 weeks	no effect on body weight or female survival;	Davison and Sell (1974)
tech DDT	mallard	16		diet for 11 months	no effect on body weight	Davison and Sell (1974)
tech DDT	mallard		16	diet for 11 months	reduced female survival (> 95%)	Davison and Sell (1974)
"DDT" (likely a mixture)	pheasant	21		diet for approximately 8 months	no effect on body weight	DeWitt (1956)
tech DDT	ring-necked pheasant		61	diet for 8 days	reduced survival (LC50)	Gill et al. (1970)

a Field-collected birds were used in this controlled laboratory experiment and small sample size was used.

LC50 – concentration that is lethal to 50% or an exposed population

LOAEL – lowest-observed-adverse-effects level

NOAEL – no-observed-adverse-effects level

Bold identifies the NOAEL and LOAEL selected as the TRVs.

BZTO104(e)030904

Table 5-40. BEHP Dietary Toxicity Studies for Birds

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
BEHP	ringed turtle dove	**1.45**		diet for 4 weeks	no effect on eggshell thickness	Peakall (1974)
BEHP	European starling	67.8		diet for 30 days	no effect on growth or food consumption	O'Shea and Stafford (1980)
BEHP	chicken		**329**	diet for 230 days	cessation of egg laying, abnormal ovaries	Ishida et al. (1982)

BEHP – bis(2-ethylhexyl) phthalate
LOAEL – lowest-observed-adverse-effects level
NOAEL – no-observed-adverse-effects level
Bold identifies the NOAEL and LOAEL selected as the TRVs.

BZTO104(e)030905

Table 5-41. TEQ Dietary Toxicity Studies for Mammals

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
TEQ	mink	**0.00000044**[a]	**0.00000224**	diet for 182 days including reproduction[b]	reduced offspring kit body weight, kit survival	Tillitt et al. (1996)
TEQ	mink		0.0000036	diet for 182 days including reproduction[b]	reduced offspring kit body weight at 3 and 6 weeks, gestation length, and kit survival	Heaton et al. (1995)
2,3,7,8-TCDD	guinea pig	0.00000065	0.0000049	diet for 90 days	reduced body weight	DeCaprio et al. (1986)
TEQ	mink	0.0000107	0.0000068	diet for 182 days including reproduction[b]	reduced maternal body weight; ingestion rate also significantly reduced	Heaton et al. (1995)
2,3,7,8-TCDD	rat	0.0000010	0.000010	diet for three generations	reduced litter size and F2 postnatal survival	Murray et al. (1979)
2,3,7,8-TCDD	guinea pig	0.0000049	0.0000285	diet for 90 days[c]	reduced survival (60%)	DeCaprio et al. (1986)
2,3,7,8-TCDD	mink	0.0000049	0.000049	diet for 125 days	reduced body weight, reduced adult survival (50%)[d]	Hochstein et al. (1998)
2,3,7,8-TCDD	mink		0.0001	IP injection for 12 days	reduced male body weight, reduced kit survival	Aulerich et al. (1988)
2,3,7,8-TCDD	rat	0.00001	0.0001	diet for 2 yrs	reduced female survival; reduced body weight	Kociba et al. (1978)
2,3,7,8-TCDD	rat		0.00032	diet for 13 weeks	reduced body weight	Van Birgelen et al. (1994)
2,3,7,8-TCDD	rat	0.00010		diet for three generations	no effect on body weight	Murray et al. (1979)
2,3,7,8-TCDD	hamster		0.500	single IP injection/gavage + 50 days untreated[e]	reduced body weight	Olson et al. (1980)
2,3,7,8-TCDD	mouse	0.19	1.37	single gavage + 30 days untreated[e]	reduced body weight	Chapman and Schiller (1985)

[a] A chronic NOAEL was extrapolated from the chronic LOAEL using a UF of 5.
[b] Mink were exposed to field-collected fish contaminated with dioxins, PCBs, and other unknown contaminants.
[c] Guinea pigs were exposed to dioxins through dietary and water exposure. The LOAEL TRV is based on the dietary exposure only.
[d] Body weight loss at LOAEL may be related to food suppression. A 20% loss in body weight was observed in the control group.
[e] Lab animals were exposed to a single exposure dose, and no intubation control group was reported.

bw – body weight
IP – intraperitoneal
LOAEL – lowest-observed-adverse-effects level
NOAEL – no-observed-adverse-effects level
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
Bold identifies the NOAEL and LOAEL selected as the TRVs

BZTO104(e)030906

Table 5-42. PCB Dietary Toxicity Studies for Mammals

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Total PCBs	mink	**0.0074**[a]	**0.037**	diet for multiple generations[b]	reduced offspring kit body weight, onset of estrus, decrease in females whelping	Restum et al. (1998)
Aroclor 1254	mink		0.074	diet for 290 days[b]	reduced offspring kit survival to 4 weeks (0%)	Hornshaw et al. (1983)
Aroclor 1254	mink	0.070	0.077	diet for 250 days[b]	reduced offspring kit body weight	Hornshaw et al. (1983)
Clophen A50	mink		0.089	diet for 18 months	reduced offspring kit growth	Brunström et al. (2001)
Aroclor 1254	mink		0.13	diet for 6 months	reduced offspring kit growth rate	Wren et al. (1987)
Total PCBs	mink		0.13	diet for 182 days during reproductive period[b]	reduced offspring kit body weight at 3 and 6 weeks, gestation length, kit survival	Heaton et al. (1995); Tillitt et al. (1996)
Aroclor 1254	mink	0.077	0.17	diet for 250 days[b]	reduced offspring kit survival at birth (0%)	Hornshaw et al. (1983)
Aroclor 1254	mink		0.22	diet for 4 and 9 months prior to giving birth	reduced number of offspring per female, decrease in offspring body weight	Ringer (1983)
Aroclor 1254	mink	0.13	0.26	diet for 4 months	reduced number of kits born alive (0% at 4 weeks)	Aulerich and Ringer (1977)
Clophen A50	mink	0.27		diet for 18 months	no effect on maternal body weight	Brunström et al. (2001)
Total PCBs	mink	0.26	0.32	diet for 182 days during reproductive period[b]	reduced maternal body weight	Heaton et al. (1995)
Aroclor 1254	mink		0.39	diet for 88 to 102 days	reduced number of kits whelped and born alive (0%)	Aulerich et al. (1985)
PCB mixture (composition not reported)	mink		0.51	diet for 66 days	reduced number of kits born alive	Jensen et al. (1977)
Aroclor 1242	mink		0.65	diet for 8 months	reproductive failure	Bleavins et al. (1980)
Aroclor 1254	mink		1.31	diet for 4 weeks	reduced body weight gain in adults	Hornshaw et al. (1986)
Aroclor 1254	mink		1.64	diet for 3 months	all whelps stillborn	Kihlstrom et al. (1992)
Aroclor 1254	mink	1.2	1.8	diet for 28 days	reduced female growth	Aulerich et al. (1986)
Clophen A50	mink		2.0	diet for 3 months	all whelps stillborn	Kihlstrom et al. (1992)
Aroclor 1254	mink	1.5	2.4	diet for 28 days	reduced male and female growth	Aulerich et al. (1986)
Aroclor 1016	mink		2.6	diet for 8 months	reduced birth weight and growth rate of offspring kits, and reduced adult female survival (75%)	Bleavins et al. (1980)
Aroclor 1232	rat	480	2,000	single IP injection + 14 days untreated	reduced body weight gain	Harris et al. (1993b)

[a] A chronic NOAEL was extrapolated from the chronic LOAEL using a UF of 5.
[b] Mink were exposed to field-collected fish contaminated with dioxins, PCBs, and other unknown contaminants.

bw – body weight
NOAEL – no-observed-adverse-effects level
LOAEL – lowest-observed-adverse-effects level
IP – intraperitoneal
PCB – polychlorinated biphenyl

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Bold identifies the NOAEL and LOAEL selected as the TRVs.

BZTO104(e)030908

Table 5-43. Antimony Dietary Toxicity Studies for Mammals

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Antimony chloride	rat		0.060	drinking water for 39 days	reduced maternal body weight	Rossi et al. (1987)
Antimony potassium tartrate	rat		0.57	drinking water for lifetime	reduced survival	Schroeder et al. (1970)
Antimony potassium tartrate	mouse	0.35		drinking water for lifetime	no effect on lifespan	Schroeder et al. (1968)
Antimony chloride	rat	0.060	0.60	drinking water for 39 days	reduced offspring body weight	Rossi et al. (1987)
Antimony potassium tartrate	mouse	0.35		drinking water for lifetime	no effect on body weight	Schroeder et al. (1968)
Antimony potassium tartrate	rat	0.57		drinking water for lifetime	no effect on body weight	Schroeder et al. (1970)
Potassium Antimony potassium tartrate	rat	5.86	43.9	drinking water for 13 weeks	reduced body weight	Poon et al. (1998)
Antimony oxide	rat	**1,489**		diet for 90 days	no effect on growth rate; small increase in body weight	Hext et al. (1999)

LOAEL – lowest-observed-adverse-effects level
NOAEL – no-observed-adverse-effects level
Bold identifies the NOAEL selected as the TRV.

BZTO104(e)030909

Table 5-44. Copper Dietary Toxicity Studies for Mammals

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Copper sulfate	mink	**18**	**26**	diet for 357 days	reduced offspring kit survival, litter mass[a]	Aulerich et al. (1982)
Copper sulfate	mink	43		diet for 153 to 657 days	no effect on body weight	Aulerich et al. (1982)
Copper sulfate	rat	137		diet for 13 weeks	100% survival	NTP (1993)
Copper sulfate	rat	67	137	diet for 13 weeks	reduced body weight	NTP (1993)
Copper sulfate	rat	93	197	diet for 2 weeks	reduced body weight	NTP (1993)
Copper sulfate	shrew	267		diet for 12 weeks	no effect on body weight	Dodds-Smith et al. (1992)
Copper sulfate	rat	305		diet for 2 weeks	100% survival	NTP (1993)
Copper sulfate	mouse	467		diet for 13 weeks	100% survival	NTP (1993)
Copper sulfate	mouse	227	467	diet for 13 weeks	reduced body weight	NTP (1993)
Copper sulfate	mouse	749		diet for 2 weeks	100% survival, no effect on growth	NTP (1993)

[a] At LOAEL, 38% kit mortality was observed. In mink from NOAEL and control groups, 12% and 19% mortality were observed, respectively.

bw – body weight

LOAEL – lowest-observed-adverse-effects level

NOAEL – no-observed-adverse-effects level

NTP – National Toxicity Program

Bold identifies the NOAEL and LOAEL selected as the TRVs.

BZTO104(e)030910

Table 5-45. Lead Dietary Toxicity Studies for Mammals

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Lead acetate	rat		0.55	drinking water for multi-generational	change in reflex development in offspring, reduced locomotor activity in adults	Reiter et al. (1975)
Triethyl lead chloride	mouse	0.5	1.5	gavage (water) for day 3 to 5 after mating	decreased frequency of implanted ova	Odenbro and Kihlstrom (1977)
Soluble lead	rat		3.3	food and drinking water for three generations	reduced offspring survival, increased number of runts	Schroeder and Mitchener (1971)
Lead acetate	rat		5.5	intubation for 18 days	altered operant behavior	Overmann (1977)
lead acetate	rat	5.5		drinking water for multiple generations	no effect on body weight	Reiter et al. (1975)
Soluble lead	mouse		6.3	food and drinking water for three generations	reduced offspring survival, increased number of runts, discontinuation of breeding	Schroeder and Mitchener (1971)
Lead acetate	rat	4.7	8.9	drinking water for 12 weeks	reduced growth of females through pregnancy and lactation	Kimmel et al. (1980)
Lead nitrate	rat	3.1	15.6	injection (IV) on day 9 or 15 of gestation, monitor for 11 or 5 days	decrease in percent body weight gain in pregnant rat	Hackett et al. (1982)
Lead acetate	rat		31.5	drinking water for 12 days	reduced growth	Hammond et al. (1990)
Lead acetate	rat	44		drinking water for gestation through nursing	no effect on maternal care (nursing, pup retrieval), offspring reflex development, maternal body weight	Zenick et al. (1979)
Lead acetate	rat	49.2		intubation for 18 days	reduced body weight	Overmann (1977)
Lead acetate	rat	**11**	**90**	diet for 2 years	reduced offspring weight, kidney damage	Azar et al. (1973)
Lead acetate	mouse		112	drinking water for 11 weeks	reduced pup weight and survival	Sharma and Kanwar (1985)
Lead acetate	rat		120	drinking water for age 3 days to age 133 days	altered operant behavior, reduced body weight	Angell and Weiss (1982)

bw – body weight
IV – intravenous
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
Bold identifies the NOAEL and LOAEL selected as the TRVs.

BZTO104(e)030911

Table 5-46. Mercury Dietary Toxicity Studies for Mammals

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Methylmercuric chloride	rat		0.0084	diet for three generations	reduced growth	Verschuuren et al. (1976)
Methylmercuric chloride	ranch mink	0.051		diet for 145 days[a]	no effect on adult survival (91% survival)	Wobeser et al. (1976a)
Methylmercuric chloride	ranch mink	0.043		diet for 145 days	no effect on juvenile growth	Wobeser et al. (1976a)
Mercury (chemical form unknown)	mink	**0.02**	**0.07**	diet for multiple generations[b]	reduced number of females whelped	Dansereau et al. (1999)
Mercury (chemical form unknown)	mink	0.07	0.12	diet for multi-generational[b]	reduced adult survival	Dansereau et al. (1999)
Methylmercuric chloride	rat	0.19		diet for three generations	no effect on survival or reproduction	Verschuuren et al. (1976)
Methylmercuric chloride	mink	0.16	0.25	diet for 93 days	reduced growth, reduced survival (60%)	Wobeser et al. (1976b)
Methylmercury	mink		0.64	diet for 2 months	reduced growth and survival (0%)	Aulerich et al. (1974)
Mercury chloride and methylmercury acetate	mouse	0.70		drinking water for lifetime	no effect on growth or survival	Schroeder and Mitchener (1975)
Methylmercury hydroxide	mouse	2.0	3.0	single injection (gestation day 8)	reduced pup survival and body weight, altered offspring avoidance behavior	Hughes and Annau (1976)

[a] Mink were exposed to field-collected fish.

[b] Mink prepared diet consisted of 40% freshwater field-collected fish and 60% prepared chicken carcasses.

bw – body weight

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

Bold identifies the NOAEL and LOAEL selected as the TRVs.

BZTO104(e)030912

Table 5-47. Selenium Dietary Toxicity Studies for Mammals

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Sodium selenite or seleniferous wheat	rat	**0.055**	**0.08**	diet for 6 weeks	reduced body weight	Halverson et al. (1966)
Sodium selenite or seleniferous wheat	rat	0.13	0.14	diet for 6 weeks	reduced survival	Halverson et al. (1966)
Selenomethionine	rat		0.16	diet for 110 days	reduced body weight	Behne et al. (1992)[a]
Selenite	rat	0.16		diet for 110 days	body weight	Behne et al. (1992)[a]
Potassium sodium selenite	rat	0.16	0.27	drinking water for 1 year	reduced pup growth, number weaned	Rosenfeld and Beath (1954)
Sodium selenite, Nano-Se, or organic selenium	rat	0.17	0.28	diet for 13 weeks	reduced body weight	Jia et al. (2005)[a]
Selenomethionine	hamster	0.36		diet for 21 days	body weight	Julius et al. (1983)
Sodium selenate	mouse	0.42		drinking water for multiple generations	no effect on survival	Schroeder and Mitchener (1971)
Sodium selenate	mouse		0.43	drinking water for multiple generations	reduced offspring survival, number of runts, breeding failure	Schroeder and Mitchener (1971)
Sodium selenate	mouse		0.43	drinking water for lifetime	reduced survival, body weight	Schroeder and Mitchener (1972)
Selenomethionine	hamster		0.76	diet for 21 days	reduced body weight	Julius et al. (1983)
Sodium selenite	hamster		0.87	single IV injection during gestation	reduced maternal body weight	Ferm et al. (1990)
Sodium selenite	hamster	1.82		single IV injection during gestation	no effect on number of abnormal litters	Ferm et al. (1990)
Sodium selenate	hamster	2.21	2.68	single IV injection during gestation	reduced maternal body weight, increase in number of abnormal litters	Ferm et al. (1990)
Sodium selenite	hamster		3.4	21days	reduced body weight	Julius et al. (1983)
Sodium selenite	hamster		5.8	21 days	reduced female survival	Julius et al. (1983)
Sodium selenate	hamster		7.11	4 days of gestation	reduced fetal body length	Ferm et al. (1990)

BZTO104(e)030913

Table 5-47. Selenium Dietary Toxicity Studies for Mammals

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Sodium selenite	hamster	7.11	7.90	4 days of gestation	increase in number of abnormal litters, reduction in fetal body length	Ferm et al. (1990)
Sodium selenate	hamster	7.90	8.69	4 days of gestation	reduced maternal body weight	Ferm et al. (1990)

[a] These studies have been added to the TRV review process (Attachment G5) since the submittal of the TRV technical memorandum to EPA as Appendix B of the Ecological PRE.

bw – body weight

IV – intravenous

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

Bold identifies the NOAEL and LOAEL selected as the TRVs.

BZTO104(e)030914

Table 5-48. Benzo(a)pyrene Dietary Toxicity Studies for Mammals

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Benzo(a)pyrene	mouse	**2.0**[a]	**10**	gavage for 10 days during gestation	reduced pup body weight, testes weight	MacKenzie and Angevine (1981)
Benzo(a)pyrene	mouse	33.3		diet for up to 115 days	no effect on survival[b]	Neal and Rigdon (1967)
Benzo(a)pyrene	mouse	199		diet for prior to or following mating	no effect on adult body weight, embryonic development, fertility, and occurrence of embryonic malformations[c]	Rigdon and Neal (1965)

[a] A chronic NOAEL was extrapolated from the chronic LOAEL using a UF of 5.

[b] Stomach tumor frequency increased at lower doses.

[c] No statistics – results not clearly presented in paper, overall conclusions taken from paper; decreased body weight attributed to decreased ingestion rat (not toxicity).

bw – body weight

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

Bold identifies the NOAEL and LOAEL selected as TRVs.

BZTO104(e)030915

Table 5-49. Total DDT and DDT Mixture Dietary Toxicity Studies for Mammals

Chemical Form	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Conditions	Endpoint Effect	Source
Technical DDT	rat	1.2		diet for two generations	reduced litter size and weight	Duby et al. (1971)
"DDT" mixture	mouse	1.3		diet for 120 days	reduced survival	Ware and Good (1967)
"DDT" mixture	mouse	**0.26**[a]	**1.3**	diet for 120 days	reduced litter size	Ware and Good (1967)
Technical DDT	rat	1.6		diet for 23 months	reduced adult survival, reduced growth, reduced viable litter size, reproductive life-span	Ottoboni (1972)
Technical DDT	rat		2.0	diet for 7.5 weeks	reduced fertility	Nickerson and Sniffen (1973)
Technical DDT	mouse	2.4		diet for 15 months	reduced adult survival, reduced litter size, litters per pair	Wolfe et al. (1979)
"DDT" mixture	rat	0.8	4.0	diet for 2 years	reduced number of young surviving to weaning	Fitzhugh (1948)
Technical DDT	rat	6.7	13.4	diet for 36 weeks	reduced litter size, mating and reproductive success	Jonsson et al. (1976)
Technical DDT	dog (beagle)	10		gavage for three generations	gestation length, fertilization, pregnancy, litter size; reduced pup survival, viability, growth, sex; lactation ability	Ottoboni et al. (1977)
Technical DDT	rat	13		diet for 37 weeks	reduced survival	Jonsson et al. (1976)
Technical DDT (DDD, DDE, DDT)	rat	16		diet for three generation	reduced adult survival, growth, fertility, viability, stillbirths, litter size, abnormalities, pup survival	Ottoboni (1969)
pp′-DDT, p,p′-DDD, p,p′-DDE	rat	21		diet for 6 weeks	reduced survival, body weight (males only)	Banerjee et al. (1996)
"DDT" mixture	mouse		37	two generations	reduced survival	Cannon and Holcomb (1968)
p,p′-DDD and p,p′-DDE	mouse		42	diet for 123 weeks	reduced male growth	Tomatis et al. (1974)
Technical DDT	mouse	9.2	46	diet for six generations	reduced lifespan, pup survival	Turusov et al. (1973)
"DDT" mixture	mouse	55		two generations	reduced body weight	Cannon and Holcomb (1968)
Technical DDT	hamster	113		diet for lifetime	reduced survival	Rossi et al. (1983)
Technical DDT	hamster		113	diet for lifetime	reduced body weight	Rossi et al. (1983)
Technical DDT	mouse		333	gavage for multiple generations	reduced adult survival, pregnancy frequency, time and fetal survival	Shabad et al. (1973)

[a] A chronic NOAEL was extrapolated from the chronic LOAEL using a UF of 5.

bw – body weight
NOAEL – no-observed-adverse-effect level
LOAEL – lowest-observed-adverse-effect level
Bold identifies the NOAEL and LOAEL selected as TRVs.

BZTO104(e)030916

Table 5-50. Summary of Reviewed BMF Studies

Chemical	BMF	Bird Species	Fish Tissue	Location	Source
Dioxins/Furans					
2,3,7,8-TCDD	2, 23	great blue heron	fish tissue regurgitated or rejected cast from nests; prey items collected opportunistically; fish species not weighted as proportion in diet	Lower Columbia River	Thomas and Anthony (1999)
2,3,7,8-TCDD	8	great blue heron	fish tissue regurgitated or rejected cast from nests; prey items collected opportunistically; fish species not weighted as proportion in diet	Willamette River	Thomas and Anthony (1999)
2,3,7,8-TCDD	**10**[a]	osprey	fish tissue composites of three resident species: largescale sucker, mountain whitefish, and northern pikeminnow	Willamette River	Henny et al. (2003)
2,3,7,8-TCDD	16 (14 – 17)	bald eagle	common carp, peamouth chub, and largescale sucker (data collected from Tetra Tech [1992])	Lower Columbia River	Buck (2004)
TEQ	19	osprey	Unknown	Michigan	Kubiak and Best (1991) as cited in Giesy et al. (1995)
2,3,7,8-TCDD	20 (15 – 30)	bald eagle	common carp, peamouth chub, and largescale sucker (data collected from Tetra Tech [1992])	Lower Columbia River	Buck (2004)
2,3,7,8-TCDD	21[b]	herring gull	alewife fish tissue	Lake Ontario	Braune and Norstrom (1989)
PCBs					
Total PCBs	5, 4, 73	great blue heron	fish tissue regurgitated or rejected cast from nests; prey items collected opportunistically; fish species not weighted as proportion in diet	Lower Columbia River	Thomas and Anthony (1999)
Total PCBs	13, 4	great blue heron	fish tissue regurgitated or rejected cast from nests; prey items collected opportunistically; fish species not weighted as proportion in diet	Willamette River	Thomas and Anthony (1999)
Total PCBs	**11**[a]	osprey	fish tissue composites of three resident species: largescale sucker, mountain whitefish, and northern pikeminnow	Willamette River	Henny et al. (2003)
Total PCBs	13	great blue heron	fish tissue regurgitated or rejected cast from nests; prey items collected opportunistically; fish species not weighted as proportion in diet	Puget Sound reference site	Thomas and Anthony (1999)
PCBs	23	brown pelican	regurgitated menhaden	South Carolina, 1973	Blus et al. (1977)
Total PCBs	28	bald eagle	various whole-body fish tissue prey items by percentage of diet	Great Lakes	Giesy et al. (1995)
Total PCBs	32[b]	herring gull eggs	alewife fish tissue	Lake Ontario	Braune and Norstrom (1989)
Total PCBs	50 (38 – 52)	bald eagle	common carp, peamouth chub, and largescale sucker (data collected from Tetra Tech [1992])	Lower Columbia River	Buck (2004)
Total PCBs	113 (90 – 155)	bald eagle	common carp, peamouth chub, and largescale sucker (data collected from Tetra Tech [1992])	Lower Columbia River	Buck (2004)
Pesticides					
p,p′-DDE	9, 3, 143	great blue	fish tissue regurgitated or rejected cast from nests; prey items collected	Lower Columbia	Thomas and Anthony

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Table 5-50. Summary of Reviewed BMF Studies

Chemical	BMF	Bird Species	Fish Tissue	Location	Source
		heron	opportunistically; fish species not weighted as proportion in diet	River	(1999)
p,p′-DDE	16, 24	great blue heron	fish tissue regurgitated or rejected cast from nests; prey items collected opportunistically; fish species not weighted as proportion in diet	Willamette River	Thomas and Anthony (1999)
p,p′-DDE	22	bald eagle	mean concentration of whole-body fish tissue by percentage of diet; concentration collected in fish and bald eagle eggs collected from inland or coastal areas	Great Lakes	Giesy et al. (1995)
DDE	31	brown pelican	menhaden fish tissue	South Carolina 1973	Blus et al. (1977)
DDE	34[b]	herring gull	alewife fish tissue	Lake Ontario	Braune and Norstrom (1989)
p,p′-DDE	41	great blue heron	fish tissue regurgitated or rejected cast from nests; prey items collected opportunistically; fish species not weighted as proportion in diet	Puget Sound reference site	Thomas and Anthony (1999)
DDE	75 (61 – 78)	bald eagle	common carp, peamouth chub, and largescale sucker	Lower Columbia River	Buck (2004)
DDE	**87[a]**	**osprey**	fish tissue composites of three resident species: largescale sucker, mountain whitefish, and northern pikeminnow	Willamette River	Henny et al. (2003)
DDE	141 (122 – 157)	bald eagle	geometric mean of common carp, peamouth chub, and largescale sucker (data collected from Tetra Tech [1992])	Lower Columbia River	Buck (2004)
Metals					
Mercury	1	bald eagle	mean concentration of whole-body fish tissue by percentage of diet	Great Lakes	Giesy et al. (1995)
Mercury	2.0 (1.7 – 2.0)	bald eagle	geometric mean of common carp, peamouth chub, and largescale sucker (data collected from Tetra Tech [1992])	Lower Columbia River	Buck (2004)
Mercury	**3.0** (1.8 – 4.6)	bald eagle	geometric mean of common carp, peamouth chub, and largescale sucker (data collected from Tetra Tech [1992])	Lower Columbia River	Buck (2004)

[a] Average lipid content (%) egg: fish = 4.3:5.0.
[b] Average lipid content (%) egg: fish = 7.7:2.8.
BMF – biomagnifications factor
PCB – polychlorinated biphenyl
TCDD – tetrachlorodibenzo-*p*-dioxin
Bold identifies BMFs selected for estimating egg exposure concentration.

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Table 5-51. Biomagnification Factors Used to Estimate Bird Egg Tissue Concentrations

Round 2 COPC	BMF	Bird Species	Location	Source
Mercury	3.0	bald eagle	Lower Columbia River	Buck (2004)
PCB TEQ	10	osprey	Willamette River	Henny et al. (2003)
Total PCBs	11	osprey	Willamette River	Henny et al. (2003)
Dioxin TEQ	10	osprey	Willamette River	Henny et al. (2003)
Sum DDE	87	osprey	Willamette River	Henny et al. (2003)

BMF – biomagnifications factor
COPC – chemical of potential concern
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

Table 5-52. Estimated Bird Egg Tissue Concentrations of Round 2 COPCs for Bald Eagle and Osprey

Round 2 COPC	Unit (ww)	Bird Egg Tissue Concentration	
		Osprey	Bald Eagle
Mercury	mg/kg	0.302	0.238
PCB TEQ	ng/kg	293	332
Total PCBs	µg/kg	19,998	40,545
Dioxin TEQ	ng/kg	136	233
Sum DDE	µg/kg	15,468	16,339

COPC – chemical of potential concern
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
ww – wet weight

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Table 5-53. 2,3,7,8-TCDD and TEQ Bird Egg Toxicity Studies

Chemical Form	Species	NOAEL (pg/g ww)	LOAEL (pg/g ww)	Exposure Conditions	Endpoint Effect	Source
2,3,7,8-TCDD	osprey	**2.3**		Willamette River, 1993	productivity	Henny et al. (2003)
2,3,7,8-TCDD	great blue heron	2.57 – 7.42		Lower Columbia River and Willamette River, 1994 to 1995	breeding behavior reproductive success, fledging	Thomas and Anthony (1999; 2003)
2,3,7,8-TCDD	chicken		10	single injection in laboratory	reduced reproductive success	Henshel et al. (1993)
TEQ	wood duck	5	20 – 50	Bayou Meto, central Arkansas wetland, 1987	nesting success, hatchability, duckling production	White and Seginak (1994)
TEQ	chicken		30	single injection in lab on day 4	reduced hatching weight without yolk sac, embryo and hatching malformation	Hoffman et al. (1998)
2,3,7,8-TCDD	bald eagle		**31.98**	Columbia River estuary, 1980 to 1987	reduced productivity, eggshell thinning	Anthony et al. (1993)
2,3,7,8-TCDD	Forster's tern		> 37.3	Green Bay, Lake Michigan during 1983 nesting season	increased incubation period, reduced hatchability, lower offspring body weight, and edema occurrence	Kubiak et al. (1989)
2,3,7,8-TCDD	osprey		78	Wisconsin River, downstream from bleach kraft mills, 1992 to 1996	reduced chick growth	Woodford et al. (1998)
TEQ	double-crested cormorant		85 – 344	five regional areas of the Great Lakes, 1986 to 1988	reduced reproductive success	Tillitt et al. (1992)
2,3,7,8-TCDD	chicken		122	single injection in laboratory	reduced embryo survival (LC50)	Henshel et al. (1997)
TEQ	osprey		136	Wisconsin River, downstream from bleach kraft mills, 1992 to 1996	reduced chick growth	Woodford et al. (1998)
2,3,7,8-TCDD	chicken		150	single injection in laboratory	embryo abnormality and reduced survival (LC50)	Powell et al. (1996)
2,3,7,8-TCDD	chicken	80	160	single injection in laboratory	reduced embryo survival, embryo deformities	Powell et al. (1996)
TEQ	chicken		230	single injection in laboratory	reduced embryo survival (LC50)	Powell et al. (1996)
2,3,7,8-TCDD	chicken	100	300	single injection in laboratory	reduced embryo survival and hatching body weight	Henshel et al. (1997)
TEQ	chicken	60	300	single injection in laboratory	reduced hatching success, embryo malformation	Hoffman et al. (1998)
TEQ	chicken	160	320	single injection in laboratory	reduced embryo survival, embryo deformities	Powell et al. (1996)
TEQ	double-crested cormorant	322		field exposure	reduced embryo survival, gross abnormalities, body weight	Powell et al. (1997)

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Table 5-53. 2,3,7,8-TCDD and TEQ Bird Egg Toxicity Studies

Chemical Form	Species	NOAEL (pg/g ww)	LOAEL (pg/g ww)	Exposure Conditions	Endpoint Effect	Source
TEQ	double-crested cormorant		350 – 1,300	Upper Great Lakes from May to August 1988	incidence of embryo malformations	Yamashita et al. (1993)
2,3,7,8-TCDD	ring-necked pheasant	100	1,000	single injection in laboratory	reduced embryo survival	Nosek et al. (1992)
2,3,7,8-TCDD	pigeon		1,000	single injection in laboratory	reduced hatchability	Janz and Bellward (1996)
TEQ	Caspian tern		1,300 – 2,800	Upper Great Lakes from May to August 1988	increased incidence of embryo malformations	Yamashita et al. (1993)
2,3,7,8-TCDD	great blue heron		2,000	single injection in laboratory	reduced hatchability	Janz and Bellward (1996)
TEQ	Forster's tern		2,175	Green Bay, Lake Michigan during 1983 nesting season	increased incubation period, reduced hatchability, lower offspring body weight, and edema occurrence	Kubiak et al. (1989)
2,3,7,8-TCDD	double-crested cormorant		4,000	single injection in laboratory	reduced embryo survival	Powell et al. (1997)
2,3,7,8-TCDD	double-crested cormorant		4,000	single injection in laboratory	reduced embryo survival	Powell et al. (1998)
TEQ	common tern		4,400	single injection on day 4 in laboratory	reduced embryo survival, hatching success	Hoffman et al. (1998)
TEQ	American kestrel		5,000	single injection on day 6	reduced embryo survival, hatching success, embryo malformation	Hoffman et al. (1998)
2,3,7,8-TCDD	double-crested cormorant	1,300	5,400	single injection in laboratory	reduced embryo survival	Powell et al. (1998)
2,3,7,8-TCDD	ring-necked pheasant	1,000	10,000	single injection in laboratory	reduced embryo survival	Nosek et al. (1992)
TEQ	double-crested cormorant		16,000	single injection in laboratory	reduced embryo survival	Powell et al. (1997)
TEQ	double-crested cormorant	7,000	17,500	single injection in laboratory	reduced embryo survival	Powell et al. (1998)
TEQ	double-crested cormorant		17,700	single injection in laboratory	reduced embryo survival	Powell et al. (1998)

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Table 5-53. 2,3,7,8-TCDD and TEQ Bird Egg Toxicity Studies

Chemical Form	Species	NOAEL (pg/g ww)	LOAEL (pg/g ww)	Exposure Conditions	Endpoint Effect	Source
TEQ	American kestrel	2,300	23,000	single injection on day 6 in laboratory	reduced hatchability	Hoffman et al. (1998)
TEQ	double-crested cormorant	20,000	40,000	single injection in laboratory	reduced embryo survival	Powell et al. (1997)

LC50 – concentration that is lethal to 50% of an exposed population
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
ww – wet weight
Bold identifies the NOAEL and LOAEL selected as the TRVs.

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Table 5-54. PCB Bird Egg Toxicity Studies

Chemical Form	Species	NOAEL (pg/g ww)	LOAEL (pg/g ww)	Exposure Conditions	Endpoint Effect	Source
Total PCBs	double-crested cormorant		4.4 – 14.8	five regional areas of the Great Lakes, 1986 to 1988	reduced reproductive success	Tillitt et al. (1992)
Total PCBs	great blue heron	0.474 – 2.842		Lower Columbia River and Willamette River, 1994 to 1995	nest attendance and visitation, fledging, reproductive success	Thomas and Anthony (1999; 2003)
Aroclor mixture	osprey	0.69		Willamette River, 1993	productivity	Henny et al. (2003)
Aroclor 1248	screech owl	3.32 and 7.12		laboratory diet for two generations	eggshell thickness, egg production, hatching success, fledging success	McLane and Hughes (1980)
Aroclor 1248	chicken	0.41	3.1	laboratory diet for 8 weeks	reduced egg production and egg hatchability	Scott et al. (1975)
Total PCBs	double-crested cormorant		3.6 – 7.3	Upper Great Lakes, May to August 1988	incidence of embryo malformations	Yamashita et al. (1993)
Total PCBs	black-crowned night heron		4.1	Bair Island, San Francisco Bay National Wildlife Refuge, 1983	reduced embryonic growth	Hoffman et al. (1986)
Total PCBs	Caspian tern		4.2 – 18	Upper Great Lakes, May to August 1988	increased incidence of embryo malformations	Yamashita et al. (1993)
Total PCBs	bald eagle	<4.5	**4.5** – < 10	14 states, 1969 to 1979	5-year productivity	Wiemeyer et al. (1984)
Total PCBs	brown pelican	4.75		South Carolina, 1969 to 1973	reproductive success	Blus et al. (1977)
Total PCBs	bald eagle	**3.0** – < 5.6	5.6 – < 13	15 states, 1980 to 1984	5-year productivity	Wiemeyer et al. (1993)
Aroclor 1248	American kestrel		5.6	laboratory diet for 5.5 months	reduced eggshell weight and thickness	Lowe and Stendell (1991)
Total PCBs	Forster's tern	7.0		Green Bay, Lake Michigan, 1988	hatching success	Harris et al. (1993a)
Total PCBs	bald eagle		12.7	Columbia River estuary, 1980 to 1987	reduced productivity, eggshell thinning	Anthony et al. (1993)
Aroclor 1254	chicken	< 5	> 15	diet for 39 weeks (14 weeks for 50 ppm group)	reduced hatchability	Platonow and Reinhart (1973)
Aroclor 1254	ringed turtle dove		16	diet for two generations	reduced hatchability, reduced embryo survival	Peakall et al. (1972); Peakall and Peakall (1973)
Total PCBs	red-breasted merganser	20		Lake Michigan, 1977 and 1978	hatching success	Haseltine et al. (1981)

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Table 5-54. PCB Bird Egg Toxicity Studies

Chemical Form	Species	NOAEL (pg/g ww)	LOAEL (pg/g ww)	Exposure Conditions	Endpoint Effect	Source
Aroclor 1254	mallard	23		laboratory diet for approximately 1 month	number hens laying, time to first hatch, clutch size, egg fertility, egg hatchability, duckling survival to 3 weeks	Custer and Heinz (1980)
Total PCBs	Forster's tern		22.2	Green Bay, Lake Michigan, during 1983 nesting season	impaired reproductive success	Kubiak et al. (1989)
Total PCBs	double-crested cormorant		23.8	Lake Huron, 1972 to 1973	impaired reproductive success	Weseloh et al. (1983)
Aroclor mixture	American kestrels		34.1	laboratory diet for 100 days until eggs hatched	impaired reproductive success of parents exposed *in ovo*	Fernie et al. (2000; 2001)
Aroclor mixture	American kestrels	34.1		laboratory diet for 1 month prior to mating through mating period	courtship behavior	Fisher et al.(2001)
Aroclor 1242	mallard	105		laboratory diet for 12 weeks	hatchability, reduced embryo survival, egg viability, embryo abnormalities	Haseltine and Prouty (1980)
PCBs	herring gulls	142		single injection in laboratory	reduced embryo and chick survival	Gilman et al. (1978)

LOAEL – lowest-observed-adverse-effects level
NOAEL – no-observed-adverse-effects level
PCB – polychlorinated biphenyl
ww – wet weight
Bold identifies the NOAEL and LOAEL selected as the TRVs.

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Table 5-55. DDE and DDT Bird Egg Toxicity Studies

Chemical Form	Species	NOAEL (pg/g ww)	LOAEL (pg/g ww)	Exposure Conditions	Endpoint Effect	Source
p,p′-DDE	great blue heron	0.331 – 1.432		Lower Columbia River and Willamette River, 1994 to 1995	nest attendance and visitation, fledging, reproductive success	Thomas and Anthony (1999; 2003)
DDE	osprey	0.82 – 22		Eagle Lake, California, 1973 to 1979 and 1983 to 1984	eggshell thinning	Littrell (1986)
DDE	clapper rail	1.3		eastern and southern United States, 1972 to 1974	eggshell thinning	Klaas et al. (1980)
DDE	brown pelicans	2.09		South Carolina, 1969 to 1973	reproductive success	Blus et al. (1977)
DDE	bald eagle	2.17 – 3.3		Pacific Coast of Canada- Fraser River (near bleached kraft paper mills) 1990 to 1992	eggshell thinning	Elliott et al. (1996)
DDE	osprey	2.3		Willamette River in 1993	productivity	Henny et al. (2003)
DDE	bald eagle	**1.3**	**3.5**	14 states, 1969 to 1979	reduced 5-year productivity	Wiemeyer et al. (1984)
DDE	bald eagle	< 2.2 – < 3.6	3.6 – < 6.3	15 states, 1980 to 1984	reduced 5-year productivity	Wiemeyer et al. (1993)
DDE	black-crowned night heron	3.9		New England and North Carolina	reproductive success	Custer et al. (1983)
DDE	double-crested cormorants	3.9		field exposure: Green Bay, Lake Michigan, 1994 and 1995	hatching success	Custer et al. (1999)
DDE	osprey	2.9	3.9	14 states, 1970 to 1979	reduced brood size	Wiemeyer et al. (1988)
DDE	common tern		3.98	Buffalo Lake, Alberta, Canada, May to August 1972	reduced reproductive success	Fox (1976)
DDE	bald eagle		4	15 states, 1980 to 1984	eggshell thinning	Wiemeyer et al. (1993)
DDE	osprey		4.2 – 8.7	14 states, 1970 to 1979	eggshell thinning	Wiemeyer et al. (1988)
DDE	long-billed curlew	4.26		Umatilla National Wildlife Refuge, 1978 to 1979	eggshell thinning	Blus et al. (1985)
DDE	cooper's hawk	4.48		South-central Ontario 1986 - 1989	eggshell thinning	Elliott and Martin (1994)
DDE	black skimmer	6.5		three sites along south Texas coast, 1979 to 1981	eggshell thinning	White et al. (1984)
DDE	black skimmer	3.2	7.0	Lavaca Bay, Texas, April to July 1984	reduced hatching success	King et al. (1991)
DDE	sharp-shinned	7.23		South-central Ontario 1986 to 1989	eggshell thinning	Elliott and Martin (1994)

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Table 5-55. DDE and DDT Bird Egg Toxicity Studies

Chemical Form	Species	NOAEL (pg/g ww)	LOAEL (pg/g ww)	Exposure Conditions	Endpoint Effect	Source
	hawk					
DDE	red-breasted merganser	7.4 – 7.6		Lake Michigan 1977 and 1978	hatching success	Haseltine et al. (1981)
DDE	peregrine falcon	9.3 and 10.6		Alaska 1960s to 1980s	reproductive success	Ambrose et al.(1988)
p,p′-DDE	bald eagle		9.7	Columbia River estuary, 1980 to 1987	eggshell thinning, reduced productivity	Anthony et al. (1993)
"DDE"	barn owls		12 - 41	laboratory diet for 2 years	eggshell breakage, reduced shell thickness, reduced nestling survival	Mendenhall et al. (1983)
DDE	double-crested cormorants		14.5	Lake Huron, 1972 to 1973	reduced reproductive success and eggshell thinning	Weseloh et al. (1983)
p,p′-DDE	mallard		15.4	laboratory diet for 96 days	reduced eggshell thickness	Haegele and Hudson (1974)
DDE	northern gannet		18.5	eastern Canada, 1968 to 1984	eggshell thinning	Elliott et al. (1988)
p,p′-DDE	bald eagle	10.72 and 4.76	21.8, 18	Maine (area with poor nesting success), 1968	reduced reproductive success	Krantz et al.(1970)
DDE	herring gull	22.6		single injection in the laboratory	reduced embryo and chick survival	Gilman et al. (1978)
p,p′-DDE	American kestrel	3.09	32.4	laboratory diet for 1 year	eggshell thickness	Wiemeyer and Porter (1970)
DDT mixture, mainly DDE	peregrine falcon		43	Norway, 1976 to 1979	eggshell thinning	Nygard (1983)
DDE	black duck		46.3	diet for reproductive period	eggshell thinning and cracking, reduced embryo survival, reduced duckling survival	Longcore et al. (1971)
commercial DDT	chicken	53 – 62		laboratory diet for 8 weeks	egg production, egg hatchability, and egg strength	Scott et al. (1975)
p,p′-DDE	black duck		64.9	laboratory diet for 7 months	reduced hell thickness, egg weight, hatchability, reduced duckling survival	Longcore and Samson (1973)
DDE mixture	American kestrel	1.9 – 7.88	84.5	laboratory diet for approximately 5 months	eggshell thickness	Lincer (1975)

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Table 5-55. DDE and DDT Bird Egg Toxicity Studies

Chemical Form	Species	NOAEL (pg/g ww)	LOAEL (pg/g ww)	Exposure Conditions	Endpoint Effect	Source
DDT	chicken	50	100	single injection	reduced chick survival	Dunachie and Fletcher (1969)
"DDT" – likely a mixture	ring-necked pheasant		162	diet for 8 weeks	offspring survival	Genelly and Rudd (1956)
p,p′-DDT	Japanese quail	105	184	single injection in the laboratory	reduced hatchability of eggs from in ovo treated birds	Bryan et al. (1989)
o,p′-DDT	Japanese quail	105	526	single injection in the laboratory	hatchability of eggs from in ovo treated birds, survival to 5 weeks	Bryan et al. (1989)
p,p′-DDT	Japanese quail	658		single injection in the laboratory	reproductive behavior	Bryan et al. (1989)
o,p′-DDT	Japanese quail	184	658	single injection in laboratory	impaired reproductive behavior	Bryan et al. (1989)

LOAEL – lowest-observed-adverse-effects level
NOAEL – no-observed-adverse-effects level
ww – wet weight
Bold identifies the NOAEL and LOAEL selected as the TRVs.

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Table 5-56. Mercury Bird Egg Toxicity Studies

Chemical Form	Species	NOAEL (µg/g ww)	LOAEL (µg/g ww)	Exposure Conditions	Endpoint Effect	Source
Mercury	bald eagle	**0.50**		14 states, 1969 to 1979	5-year productivity	Wiemeyer et al. (1984)
Mercury	red-breasted merganser	0.52		Lake Michigan, 1977 and 1978	hatching success	Haseltine et al. 1981
Mercuric chloride	Japanese quail (1 day old)	0.30	0.7	laboratory diet for 10 weeks	eggshell thickness	Stoewsand et al. (1971)
Methylmercury chloride	mallard		**0.74** – 38	laboratory diet for reproductive period	impaired embryo development	Heinz and Hoffman (2003)
Methylmercury	mallard		0.83	laboratory diet for three generations	reduced egg and young production, eggshell thinning, duckling response to maternal call, duckling avoidance behavior	Heinz (1979)
Methylmercury chloride	mallard		0.93 – 18	laboratory diet for reproductive period	impaired embryo development	Heinz and Hoffman (2003)
Methylmercury	mallard	0.79 – 1.05	5.46 – 7.39	laboratory diet for 2 years	reduced offspring survival, avoidance response of offspring	Heinz (1976)
Methylmercury dicyandiamide	mallard	1.0	6.0 – 9.0	laboratory diet for through hatching	depressed hatching success, depressed egg production, and reduced duckling survival to 1 week of age	Heinz (1974)
Mercuric sulfate	chicken		29.0	laboratory diet for 8 weeks	reduced hatchability	Scott et al. (1975)
Methylmercury chloride	chicken		40.2	laboratory diet for 8 weeks	reduced egg weight, fertility, hatchability of eggs	Scott et al. (1975)

LOAEL – lowest-observed-adverse-effects level
NOAEL – no-observed-adverse-effects level
ww – wet weight
Bold identifies the NOAEL and LOAEL selected as the TRVs.

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Table 5-57. HQ Calculations for Spotted Sandpiper Based on the Study Area-Wide Assumption for the Dietary Dose Approach

Round 2 COPC	Unit (bw/day)	TRVs		Clam Diet					Worm Diet				
					HQ		Contribution to Risk (%)			HQ		Contribution to Risk (%)	
		NOAEL	LOAEL	IR_{diet}	NOAEL	LOAEL	Tissue	Sediment	IR_{diet}	NOAEL	LOAEL	Tissue	Sediment
Arsenic	mg/kg	2.3	6.8	1.23	0.53	0.18	92.1%	7.9%	1.73	0.75	0.26	94.4%	5.6%
Cadmium	mg/kg	0.73	2.9	0.152	0.21	0.052	84.6%	15.4%	0.121	0.17	0.042	80.8%	19.2%
Chromium	mg/kg	1	5	1.62	**1.6**	0.32	49.6%	50.4%	1.48	**1.5**	0.30	44.8%	55.2%
Copper	mg/kg	47	62	13.3	0.28	0.22	86.5%	13.5%	4.71	0.10	0.076	61.9%	38.1%
Lead	mg/kg	2	20	1.79	0.89	0.089	8.5%	991.5%	2.10	**1.1**	0.11	22.2%	77.8%
Mercury	mg/kg	0.010	0.05	0.0144	**1.4**	0.29	80.8%	19.2%	0.0100	1.0	0.20	72.3%	27.7%
Selenium	mg/kg	0.42	0.82	0.154	0.37	0.19	98.2%	1.8%	0.341	0.82	0.42	992%	0.8%
Thallium	mg/kg	0.48	24	0.442[a]	**0.92**	0.018	0.2%	99.8%	NA	**NA**	NA	NA	NA
Zinc	mg/kg	82	124	51.3	0.63	0.42	84%	16%	39.7	0.48	0.32	79.4%	20.6%
Benzo(a)pyrene	µg/kg	280	1,400	244	0.87	0.17	95.7%	4.3%	139	0.50	0.099	92.5%	7.5%
Total PAHs	µg/kg	40,000	NA	885	0.022	NA	92.1%	7.9%	14,100	0.35	NA	99.5%	0.5%
BEHP	µg/kg	1,450	329,000	101	0.069	0.00031	95.0%	5.0%	157	0.11	0.00048	96.8%	3.2%
Dibutyl phthalate	µg/kg	1,450	329,000	671	0.46	0.0020	99.1%	0.9%	309	0.21	0.00094	98.0%	2.0%
PCB TEQ	pg/g	14	140	38.2	**2.7**	0.27	79.4%	20.6%	440	**31**	**3.1**	98.2%	1.8%
Total PCBs	µg/kg	290	580	742	**2.6**	**1.3**	97.7%	2.3%	3,270	**11**	**5.6**	99.5%	0.5%
Dioxin TEQ	pg/g	14	140	13.2	0.95	0.095	92.5%	7.5%	423	**30**	**3.0**	99.8%	0.2%
Aldrin	µg/kg	8	40	1.31	0.16	0.033	98.0%	2.0%	16.3	**2.0**	0.41	99.8%	0.2%
Sum DDD	µg/kg	180	900	153	0.85	0.17	98.9%	1.1%	714	**4.0**	0.79	99.8%	0.2%
Sum DDE	µg/kg	64	320	45.6	0.71	0.14	96.4%	3.6%	78.6	**1.2**	0.25	97.9%	2.1%
Sum DDT	µg/kg	30	150	67.6	**2.3**	0.45	94.9%	5.1%	27.6	0.92	0.18	87.4%	12.6%
Total DDTs	µg/kg	180	1800	168	0.94	0.094	96.3%	3.7%	852	**4.7**	0.47	99.3%	0.7%

[a] Thallium dose estimated based on clam diet only using a sediment UCL based on all sediment samples collected in the Study Area (thallium was not analyzed in beach sediment samples or in laboratory-tested worms).

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – low-observed-adverse-effect level
NA – not available
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

Bold identifies HQs > 1.0.

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Table 5-58. HQ Calculations for Spotted Sandpiper Based on the Dietary Dose Approach Evaluated at Every Shorebird Beach Habitat Area

Area	IR_{dose}		TRVs		HQs Based on Clam Diet		HQs Based on Worm Diet	
	Clam	Worm	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Arsenic (mg/kg bw/day)								
B1	1.22	1.74	2.3	6.8	0.53	0.18	0.76	0.26
B2	1.31	1.54	2.3	6.8	0.57	0.19	0.67	0.23
B3	1.54	1.03	2.3	6.8	0.67	0.23	0.45	0.15
B4	1.20	1.71	2.3	6.8	0.52	0.18	0.75	0.25
B5	1.30	0.648	2.3	6.8	0.57	0.19	0.28	0.096
B6	1.20	2.53	2.3	6.8	0.52	0.18	**1.1**	0.37
B7	1.25	1.76	2.3	6.8	0.54	0.18	0.77	0.26
B8	1.27	1.78	2.3	6.8	0.55	0.19	0.78	0.26
B9	1.25	2.14	2.3	6.8	0.54	0.18	0.93	0.32
B10	1.13	1.62	2.3	6.8	0.49	0.17	0.70	0.24
B11	0.555	2.24	2.3	6.8	0.24	0.082	0.97	0.33
B12	1.28	0.771	2.3	6.8	0.55	0.19	0.34	0.11
B13	1.04	0.982	2.3	6.8	0.45	0.15	0.43	0.14
B14	1.21	1.73	2.3	6.8	0.53	0.18	0.75	0.25
B15	1.21	2.04	2.3	6.8	0.53	0.18	0.89	0.30
B16	1.32	1.42	2.3	6.8	0.57	0.20	0.62	0.21
B17	1.25	2.16	2.3	6.8	0.54	0.18	0.94	0.32
B18	1.15	3.64	2.3	6.8	0.50	0.17	**1.6**	0.54
B19	1.22	2.39	2.3	6.8	0.53	0.18	1.0	0.35
B20	1.24	1.30	2.3	6.8	0.54	0.18	0.57	0.19
B21	0.991	1.17	2.3	6.8	0.43	0.15	0.51	0.17
B22	1.33	2.11	2.3	6.8	0.58	0.20	0.92	0.31
B23	1.07	1.68	2.3	6.8	0.46	0.16	0.73	0.25
B24	1.14	1.11	2.3	6.8	0.50	0.17	0.48	0.16
B25	1.27	1.65	2.3	6.8	0.55	0.19	0.72	0.24
B26	1.13	1.06	2.3	6.8	0.49	0.17	0.46	0.16
B27	0.547	1.22	2.3	6.8	0.24	0.081	0.53	0.18
B28	0.678	2.15	2.3	6.8	0.29	0.10	0.94	0.32
Cadmium (mg/kg bw/day)								
B1	0.103	0.102	0.73	2.9	0.14	0.036	0.14	0.035
B2	0.146	0.0794	0.73	2.9	0.20	0.051	0.11	0.027
B3	0.278	0.290	0.73	2.9	0.38	0.096	0.40	0.10
B4	0.0952	0.100	0.73	2.9	0.13	0.033	0.14	0.035
B5	0.141	0.122	0.73	2.9	0.19	0.049	0.17	0.042
B6	0.158	0.129	0.73	2.9	0.22	0.055	0.18	0.045
B7	0.0905	0.0988	0.73	2.9	0.12	0.031	0.14	0.034
B8	0.0938	0.100	0.73	2.9	0.13	0.032	0.14	0.035
B9	0.187	0.0651	0.73	2.9	0.26	0.065	0.089	0.023

Table 5-58. HQ Calculations for Spotted Sandpiper Based on the Dietary Dose Approach Evaluated at Every Shorebird Beach Habitat Area

Area	IR_{dose}		TRVs		HQs Based on Clam Diet		HQs Based on Worm Diet	
	Clam	Worm	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
B10	0.115	0.0779	0.73	2.9	0.16	0.040	0.11	0.027
B11	0.0583	0.0563	0.73	2.9	0.080	0.020	0.077	0.019
B12	0.0907	0.0902	0.73	2.9	0.12	0.031	0.12	0.031
B13	0.0828	0.0632	0.73	2.9	0.11	0.029	0.086	0.022
B14	0.0946	0.100	0.73	2.9	0.13	0.033	0.14	0.035
B15	0.0752	0.0689	0.73	2.9	0.10	0.026	0.094	0.024
B16	0.0940	0.0618	0.73	2.9	0.13	0.033	0.084	0.021
B17	0.0946	0.0550	0.73	2.9	0.13	0.033	0.075	0.019
B18	0.0914	0.0813	0.73	2.9	0.12	0.032	0.11	0.028
B19	0.0868	0.0560	0.73	2.9	0.12	0.030	0.077	0.019
B20	0.263	0.199	0.73	2.9	0.36	0.091	0.27	0.069
B21	0.117	0.0533	0.73	2.9	0.16	0.041	0.073	0.018
B22	0.125	0.0738	0.73	2.9	0.17	0.043	0.10	0.026
B23	0.0757	0.0504	0.73	2.9	0.10	0.026	0.069	0.017
B24	0.172	0.0985	0.73	2.9	0.23	0.060	0.13	0.034
B25	0.0784	0.0466	0.73	2.9	0.11	0.027	0.064	0.016
B26	0.0821	0.0701	0.73	2.9	0.11	0.028	0.096	0.024
B27	0.0637	0.0500	0.73	2.9	0.087	0.022	0.068	0.017
B28	0.0718	0.0607	0.73	2.9	0.10	0.025	0.083	0.021
Chromium (mg/kg bw/day)								
B1	1.57	1.40	1	5	**1.6**	0.31	**1.4**	0.28
B2	2.27	2.41	1	5	**2.3**	0.45	**2.4**	0.48
B3	3.55	3.41	1	5	**3.5**	0.71	**3.4**	0.68
B4	1.26	1.09	1	5	**1.3**	0.25	**1.1**	0.22
B5	1.48	1.16	1	5	**1.5**	0.30	**1.2**	0.23
B6	1.29	0.827	1	5	**1.3**	0.26	0.83	0.17
B7	1.29	1.13	1	5	**1.3**	0.26	**1.1**	0.23
B8	1.57	1.41	1	5	**1.6**	0.31	**1.4**	0.28
B9	1.17	1.31	1	5	**1.2**	0.23	**1.3**	0.26
B10	1.23	0.922	1	5	**1.2**	0.25	0.92	0.18
B11	0.598	0.721	1	5	0.60	0.12	0.72	0.14
B12	1.24	0.888	1	5	**1.2**	0.25	0.89	0.18
B13	1.47	1.77	1	5	**1.5**	0.29	**1.8**	0.35
B14	1.17	1.01	1	5	**1.2**	0.23	1.0	0.20
B15	1.38	1.18	1	5	**1.4**	0.28	**1.2**	0.24
B16	2.09	2.12	1	5	**2.1**	0.42	**2.1**	0.42
B17	2.73	2.65	1	5	**2.7**	0.55	**2.6**	0.53
B18	1.40	0.953	1	5	**1.4**	0.28	0.95	0.19
B19	1.18	1.15	1	5	**1.2**	0.24	**1.2**	0.23

Table 5-58. HQ Calculations for Spotted Sandpiper Based on the Dietary Dose Approach Evaluated at Every Shorebird Beach Habitat Area

Area	IR_{dose}		TRVs		HQs Based on Clam Diet		HQs Based on Worm Diet	
	Clam	Worm	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
B20	1.55	1.50	1	5	**1.5**	0.31	**1.5**	0.30
B21	1.26	1.02	1	5	**1.3**	0.25	1.0	0.20
B22	1.55	1.53	1	5	**1.6**	0.31	**1.5**	0.31
B23	1.29	1.38	1	5	**1.3**	0.26	**1.4**	0.28
B24	1.23	1.52	1	5	**1.2**	0.25	**1.5**	0.30
B25	1.51	1.22	1	5	**1.5**	0.30	**1.2**	0.24
B26	1.54	0.987	1	5	**1.5**	0.31	0.99	0.20
B27	1.24	1.39	1	5	**1.2**	0.25	**1.4**	0.28
B28	1.25	1.39	1	5	**1.2**	0.25	**1.4**	0.28
Copper (mg/kg bw/day)								
B1	12.3	5.06	47	62	0.26	0.20	0.11	0.082
B2	11.7	3.24	47	62	0.25	0.19	0.069	0.053
B3	13.9	3.90	47	62	0.29	0.22	0.083	0.063
B4	12.0	4.80	47	62	0.26	0.20	0.10	0.078
B5	11.6	4.08	47	62	0.25	0.19	0.087	0.066
B6	12.6	3.90	47	62	0.27	0.20	0.083	0.063
B7	12.2	4.93	47	62	0.26	0.20	0.11	0.080
B8	17.8	10.5	47	62	0.38	0.29	0.22	0.17
B9	12.9	2.73	47	62	0.27	0.21	0.058	0.044
B10	9.76	3.16	47	62	0.21	0.16	0.067	0.051
B11	4.45	3.10	47	62	0.095	0.072	0.066	0.050
B12	12.2	4.06	47	62	0.26	0.20	0.086	0.066
B13	14.1	7.47	47	62	0.30	0.23	0.16	0.12
B14	12.2	4.95	47	62	0.26	0.20	0.11	0.080
B15	14.2	3.76	47	62	0.30	0.23	0.080	0.061
B16	14.2	3.22	47	62	0.30	0.23	0.068	0.052
B17	14.5	6.09	47	62	0.31	0.23	0.13	0.099
B18	13.3	4.34	47	62	0.28	0.22	0.092	0.070
B19	11.6	3.09	47	62	0.25	0.19	0.066	0.050
B20	12.5	4.20	47	62	0.27	0.20	0.089	0.068
B21	12.1	4.90	47	62	0.26	0.20	0.10	0.079
B22	12.7	4.44	47	62	0.27	0.21	0.094	0.072
B23	10.6	3.89	47	62	0.23	0.17	0.083	0.063
B24	11.7	4.57	47	62	0.25	0.19	0.097	0.074
B25	11.7	2.81	47	62	0.25	0.19	0.060	0.045
B26	11.1	3.42	47	62	0.24	0.18	0.073	0.055
B27	5.06	3.12	47	62	0.11	0.082	0.066	0.051
B28	6.15	2.91	47	62	0.13	0.10	0.062	0.047

Table 5-58. HQ Calculations for Spotted Sandpiper Based on the Dietary Dose Approach Evaluated at Every Shorebird Beach Habitat Area

Area	IR_{dose}		TRVs		HQs Based on Clam Diet		HQs Based on Worm Diet	
	Clam	Worm	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Lead (mg/kg bw/day)								
B1	0.440	0.662	2	20	0.22	0.022	0.33	0.033
B2	0.290	0.519	2	20	0.14	0.014	0.26	0.026
B3	1.45	1.97	2	20	0.73	0.073	0.99	0.099
B4	0.206	0.388	2	20	0.10	0.010	0.19	0.019
B5	0.383	0.590	2	20	0.19	0.019	0.29	0.029
B6	0.441	0.616	2	20	0.22	0.022	0.31	0.031
B7	0.471	0.698	2	20	0.24	0.024	0.35	0.035
B8	0.443	0.666	2	20	0.22	0.022	0.33	0.033
B9	1.36	2.04	2	20	0.68	0.068	1.0	0.10
B10	0.463	1.13	2	20	0.23	0.023	0.56	0.056
B11	0.599	0.755	2	20	0.30	0.030	0.38	0.038
B12	0.517	0.711	2	20	0.26	0.026	0.36	0.036
B13	2.03	2.42	2	20	1.0	0.10	**1.2**	0.12
B14	0.659	0.920	2	20	0.33	0.033	0.46	0.046
B15	1.08	1.26	2	20	0.54	0.054	0.63	0.063
B16	0.782	1.08	2	20	0.39	0.039	0.54	0.054
B17	1.77	1.74	2	20	0.89	0.089	0.87	0.087
B18	0.535	0.834	2	20	0.27	0.027	0.42	0.042
B19	0.318	0.601	2	20	0.16	0.016	0.30	0.030
B20	7.77	8.17	2	20	**3.9**	0.39	**4.1**	0.41
B21	0.610	0.712	2	20	0.31	0.031	0.36	0.036
B22	2.43	3.28	2	20	**1.2**	0.12	**1.6**	0.16
B23	0.503	0.875	2	20	0.25	0.025	0.44	0.044
B24	0.601	0.891	2	20	0.30	0.030	0.45	0.045
B25	0.434	0.555	2	20	0.22	0.022	0.28	0.028
B26	0.786	1.00	2	20	0.39	0.039	0.50	0.050
B27	1.03	1.29	2	20	0.52	0.052	0.65	0.065
B28	0.341	0.643	2	20	0.17	0.017	0.32	0.032
Mercury (mg/kg bw/day)								
B1	0.0130	0.00833	0.01	0.05	**1.3**	0.26	0.83	0.17
B2	0.00779	0.0101	0.01	0.05	0.78	0.16	1.0	0.20
B3	0.0106	0.00523	0.01	0.05	**1.1**	0.21	0.52	0.10
B4	0.0119	0.00726	0.01	0.05	**1.2**	0.24	0.73	0.15
B5	0.0101	0.00476	0.01	0.05	1.0	0.20	0.48	0.095
B6	0.0101	0.00697	0.01	0.05	1.0	0.20	0.70	0.14
B7	0.0125	0.00779	0.01	0.05	**1.2**	0.25	0.78	0.16
B8	0.0125	0.00779	0.01	0.05	**1.2**	0.25	0.78	0.16
B9	0.0143	0.00845	0.01	0.05	**1.4**	0.29	0.84	0.17

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Table 5-58. HQ Calculations for Spotted Sandpiper Based on the Dietary Dose Approach Evaluated at Every Shorebird Beach Habitat Area

Area	IR_{dose}		TRVs		HQs Based on Clam Diet		HQs Based on Worm Diet	
	Clam	Worm	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
B10	0.0133	0.00593	0.01	0.05	**1.3**	0.27	0.59	0.12
B11	0.0116	0.00992	0.01	0.05	**1.2**	0.23	0.99	0.20
B12	0.0111	0.00553	0.01	0.05	**1.1**	0.22	0.55	0.11
B13	0.0125	0.00546	0.01	0.05	**1.2**	0.25	0.55	0.11
B14	0.0246	0.0200	0.01	0.05	**2.5**	0.49	**2.0**	0.40
B15	0.0148	0.0130	0.01	0.05	**1.5**	0.30	**1.3**	0.26
B16	0.0159	0.00770	0.01	0.05	**1.6**	0.32	0.77	0.15
B17	0.0136	0.00811	0.01	0.05	**1.4**	0.27	0.81	0.16
B18	0.0130	0.00758	0.01	0.05	**1.3**	0.26	0.76	0.15
B19	0.0122	0.00521	0.01	0.05	**1.2**	0.24	0.52	0.10
B20	0.0136	0.00897	0.01	0.05	**1.4**	0.27	0.90	0.18
B21	0.0136	0.00546	0.01	0.05	**1.4**	0.27	0.55	0.11
B22	0.0160	0.0149	0.01	0.05	**1.6**	0.32	**1.5**	0.30
B23	0.00981	0.00865	0.01	0.05	0.98	0.20	0.86	0.17
B24	0.0157	0.00612	0.01	0.05	**1.6**	0.31	0.61	0.12
B25	0.0148	0.00662	0.01	0.05	**1.5**	0.30	0.66	0.13
B26	0.0116	0.00808	0.01	0.05	**1.2**	0.23	0.81	0.16
B27	0.0129	0.00615	0.01	0.05	**1.3**	0.26	0.61	0.12
B28	0.0151	0.0113	0.01	0.05	**1.5**	0.30	**1.1**	0.23
Selenium (mg/kg bw/day)								
B1	0.154	0.341	0.42	0.82	0.37	0.19	0.82	0.42
B2	0.115	0.374	0.415	0.815	0.28	0.14	0.90	0.46
B3	0.204	0.433	0.415	0.815	0.49	0.25	1.0	0.53
B4	0.153	0.339	0.415	0.815	0.37	0.19	0.82	0.42
B5	0.138	0.413	0.415	0.815	0.33	0.17	1.0	0.51
B6	0.147	0.365	0.415	0.815	0.35	0.18	0.88	0.45
B7	0.155	0.342	0.415	0.815	0.37	0.19	0.82	0.42
B8	0.155	0.342	0.415	0.815	0.37	0.19	0.82	0.42
B9	0.102	0.100	0.415	0.815	0.25	0.13	0.24	0.12
B10	0.115	0.316	0.415	0.815	0.28	0.14	0.76	0.39
B11	0.203	0.363	0.415	0.815	0.49	0.25	0.87	0.45
B12	0.106	0.365	0.415	0.815	0.26	0.13	0.88	0.45
B13	0.202	0.108	0.415	0.815	0.49	0.25	0.26	0.13
B14	0.152	0.339	0.415	0.815	0.37	0.19	0.82	0.42
B15	0.143	0.424	0.415	0.815	0.35	0.18	1.0	0.52
B16	0.120	0.108	0.415	0.815	0.29	0.15	0.26	0.13
B17	0.200	0.353	0.415	0.815	0.48	0.25	0.85	0.43
B18	0.166	0.353	0.415	0.815	0.40	0.20	0.85	0.43
B19	0.104	0.118	0.415	0.815	0.25	0.13	0.28	0.14
B20	0.173	0.131	0.415	0.815	0.42	0.21	0.31	0.16

Table 5-58. HQ Calculations for Spotted Sandpiper Based on the Dietary Dose Approach Evaluated at Every Shorebird Beach Habitat Area

Area	IR_{dose}		TRVs		HQs Based on Clam Diet		HQs Based on Worm Diet	
	Clam	Worm	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
B21	0.108	0.132	0.415	0.815	0.26	0.13	0.32	0.16
B22	0.162	0.131	0.415	0.815	0.39	0.20	0.32	0.16
B23	0.117	0.246	0.415	0.815	0.28	0.14	0.59	0.30
B24	0.248	0.108	0.415	0.815	0.60	0.30	0.26	0.13
B25	0.106	0.132	0.415	0.815	0.26	0.13	0.32	0.16
B26	0.129	0.223	0.415	0.815	0.31	0.16	0.54	0.27
B27	0.0952	0.0940	0.415	0.815	0.23	0.12	0.23	0.12
B28	0.292	0.307	0.415	0.815	0.70	0.36	0.74	0.38
Zinc (mg/kg bw/day)								
B1	45.7	32.0	82	124	0.56	0.37	0.39	0.26
B2	40.9	34.5	82	124	0.50	0.33	0.42	0.28
B3	54.8	43.5	82	124	0.67	0.44	0.53	0.35
B4	45.9	32.4	82	124	0.56	0.37	0.39	0.26
B5	37.9	32.8	82	124	0.46	0.31	0.40	0.27
B6	58.1	31.8	82	124	0.71	0.47	0.39	0.26
B7	45.2	31.3	82	124	0.55	0.37	0.38	0.25
B8	46.2	32.8	82	124	0.56	0.37	0.40	0.27
B9	37.0	30.4	82	124	0.45	0.30	0.37	0.25
B10	41.0	29.9	82	124	0.50	0.33	0.37	0.24
B11	14.9	28.1	82	124	0.18	0.12	0.34	0.23
B12	48.1	39.5	82	124	0.59	0.39	0.48	0.32
B13	39.7	35.7	82	124	0.48	0.32	0.44	0.29
B14	44.9	30.9	82	124	0.55	0.36	0.38	0.25
B15	35.6	33.4	82	124	0.43	0.29	0.41	0.27
B16	45.6	37.9	82	124	0.56	0.37	0.46	0.31
B17	51.0	31.5	82	124	0.62	0.41	0.38	0.25
B18	35.7	31.9	82	124	0.43	0.29	0.39	0.26
B19	38.5	34.9	82	124	0.47	0.31	0.43	0.28
B20	93.4	70.5	82	124	**1.1**	0.76	0.86	0.57
B21	52.6	32.4	82	124	0.64	0.43	0.39	0.26
B22	55.4	39.3	82	124	0.68	0.45	0.48	0.32
B23	40.2	30.8	82	124	0.49	0.33	0.38	0.25
B24	40.4	37.2	82	124	0.49	0.33	0.45	0.30
B25	48.5	31.8	82	124	0.59	0.39	0.39	0.26
B26	50.3	35.2	82	124	0.61	0.41	0.43	0.29
B27	27.1	45.2	82	124	0.33	0.22	0.55	0.37
B28	19.8	29.1	82	124	0.24	0.16	0.36	0.24
TBT (µg/kg bw/day)								
B1	nd	nd	1,400	3,600	nd	nd	nd	nd
B2	6.01	0.751	1,400	3,600	0.0043	0.0017	0.00054	0.00021

Table 5-58. HQ Calculations for Spotted Sandpiper Based on the Dietary Dose Approach Evaluated at Every Shorebird Beach Habitat Area

Area	IR_{dose}		TRVs		HQs Based on Clam Diet		HQs Based on Worm Diet	
	Clam	Worm	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
B3	7.94	0.817	1,400	3,600	0.0057	0.0022	0.00058	0.00023
B4	nd	nd	1,400	3,600	nd	nd	nd	nd
B5	5.56	3.11	1,400	3,600	0.0040	0.0015	0.0022	0.00086
B6	74.7	24.5	1,400	3,600	0.053	0.021	0.017	0.0068
B7	nd	nd	1,400	3,600	nd	nd	nd	nd
B8	nd	nd	1,400	3,600	nd	nd	nd	nd
B9	9.00	4.22	1,400	3,600	0.0064	0.0025	0.0030	0.0012
B10	10.7	3.25	1,400	3,600	0.0077	0.0030	0.0023	0.00090
B11	0.884	0.779	1,400	3,600	0.00063	0.00025	0.00056	0.00022
B12	4.90	0.483	1,400	3,600	0.0035	0.0014	0.00035	0.00013
B13	11.1	3.84	1,400	3,600	0.0079	0.0031	0.0027	0.0011
B14	nd	nd	1,400	3,600	nd	nd	nd	nd
B15	9.41	4.04	1,400	3,600	0.0067	0.0026	0.0029	0.0011
B16	2.72	1.37	1,400	3,600	0.0019	0.00075	0.00098	0.00038
B17	5.44	1.59	1,400	3,600	0.0039	0.0015	0.0011	0.00044
B18	70.1	59.0	1,400	3,600	0.050	0.019	0.042	0.016
B19	4.60	1.98	1,400	3,600	0.0033	0.0013	0.0014	0.00055
B20	3.93	1.48	1,400	3,600	0.0028	0.0011	0.0011	0.00041
B21	43.9	62.6	1,400	3,600	0.031	0.012	0.045	0.017
B22	1.89	1.37	1,400	3,600	0.0014	0.00053	0.00098	0.00038
B23	5.03	9.47	1,400	3,600	0.0036	0.0014	0.0068	0.0026
B24	17.6	30.4	1,400	3,600	0.013	0.0049	0.022	0.0084
B25	1.16	0.989	1,400	3,600	0.00083	0.00032	0.00071	0.00027
B26	1.52	0.489	1,400	3,600	0.0012	0.00048	0.00050	0.00019
B27	0.956	1.77	1,400	3,600	0.00068	0.00027	0.0013	0.00049
B28	0.745	0.616	1,400	3,600	0.00053	0.00021	0.00044	0.00017
Benzo(a)pyrene (µg/kg bw/day)								
B1	11.3	11.3	280	1,400	0.040	0.0081	0.041	0.0081
B2	5.70	10.7	280	1,400	0.020	0.0041	0.038	0.0077
B3	11.6	14.0	280	1,400	0.041	0.0083	0.050	0.010
B4	0.0415	0.0444	280	1,400	0.00015	0.000030	0.00016	0.000032
B5	17.0	29.9	280	1,400	0.061	0.012	0.11	0.021
B6	18.9	43.5	280	1,400	0.068	0.014	0.16	0.031
B7	0.165	0.170	280	1,400	0.00059	0.00012	0.00061	0.00012
B8	11.3	11.3	280	1,400	0.040	0.0081	0.041	0.0081
B9	39.7	121	280	1,400	0.14	0.028	0.43	0.086
B10	9.30	28.3	280	1,400	0.033	0.0066	0.10	0.020
B11	4.62	47.8	280	1,400	0.017	0.0033	0.17	0.034
B12	46.9	1,134	280	1,400	0.17	0.033	**4.1**	0.81
B13	8.07	35.4	280	1,400	0.029	0.0058	0.13	0.025
B14	5.66	5.68	280	1,400	0.020	0.0040	0.020	0.0041

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Table 5-58. HQ Calculations for Spotted Sandpiper Based on the Dietary Dose Approach Evaluated at Every Shorebird Beach Habitat Area

Area	IR_{dose}		TRVs		HQs Based on Clam Diet		HQs Based on Worm Diet	
	Clam	Worm	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
B15	19.4	21.2	280	1,400	0.069	0.014	0.076	0.015
B16	71.7	165	280	1,400	0.26	0.051	0.60	0.12
B17	22.7	117	280	1,400	0.081	0.016	0.42	0.084
B18	3.27	8.18	280	1,400	0.012	0.0023	0.029	0.0058
B19	3.33	14.1	280	1,400	0.012	0.0024	0.050	0.010
B20	43.1	80.6	280	1,400	0.15	0.031	0.29	0.058
B21	3.16	19.2	280	1,400	0.011	0.0023	0.068	0.014
B22	3.73	15.9	280	1,400	0.013	0.0027	0.057	0.011
B23	1.83	4.29	280	1,400	0.0065	0.0013	0.015	0.0031
B24	5.06	11.8	280	1,400	0.018	0.0036	0.042	0.0084
B25	2.13	2.60	280	1,400	0.0076	0.0015	0.0093	0.0019
B26	2.36	20.0	280	1,400	0.0084	0.0017	0.071	0.014
B27	8.42	13.0	280	1,400	0.030	0.0060	0.046	0.0093
B28	2.38	4.60	280	1,400	0.0085	0.0017	0.016	0.0033
Total PAHs (µg/kg bw/day)								
B1	98.4	18,800	40,000	na	0.0025	ne	0.470	ne
B2	353	386	40,000	na	0.0088	ne	0.0096	ne
B3	429	471	40,000	na	0.011	ne	0.012	ne
B4	0.843	18,700	40,000	na	0.000021	ne	0.47	ne
B5	661	621	40,000	na	0.017	ne	0.016	ne
B6	579	1,160	40,000	na	0.014	ne	0.029	ne
B7	4.21	18,700	40,000	na	0.00011	ne	0.47	ne
B8	113	18,800	40,000	na	0.0028	ne	0.47	ne
B9	579	2,380	40,000	na	0.014	ne	0.059	ne
B10	326	1,530	40,000	na	0.008.1	ne	0.038	ne
B11	87.3	1,440	40,000	na	0.0022	ne	0.036	ne
B12	1,200	27,200	40,000	na	0.030	ne	0.68	ne
B13	221	1,640	40,000	na	0.0055	ne	0.041	ne
B14	58.1	18,700	40,000	na	0.0015	ne	0.47	ne
B15	396	651	40,000	na	0.0099	ne	0.016	ne
B16	1,400	5,920	40,000	na	0.035	ne	0.15	ne
B17	675	5,940	40,000	na	0.017	ne	0.15	ne
B18	169	371	40,000	na	0.0042	ne	0.0093	ne
B19	160	789	40,000	na	0.0040	ne	0.020	ne
B20	541	1,540	40,000	na	0.014	ne	0.039	ne
B21	122	542	40,000	na	0.0030	ne	0.014	ne
B22	180	1,340	40,000	na	0.0045	ne	0.033	ne
B23	42.9	205	40,000	na	0.0011	ne	0.0051	ne
B24	298	596	40,000	na	0.0074	ne	0.015	ne

Table 5-58. HQ Calculations for Spotted Sandpiper Based on the Dietary Dose Approach Evaluated at Every Shorebird Beach Habitat Area

Area	IR_{dose}		TRVs		HQs Based on Clam Diet		HQs Based on Worm Diet	
	Clam	Worm	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
B25	61.2	101	40,000	na	0.0015	ne	0.0025	ne
B26	131	1,290	40,000	na	0.0033	ne	0.032	ne
B27	128	460	40,000	na	0.0032	ne	0.011	ne
B28	74.9	177	40,000	na	0.0019	ne	0.0044	ne
BEHP (µg/kg bw/day)								
B1	89.0	164	1,450	329,000	0.061	0.00027	0.11	0.00050
B2	31.0	164	1,450	329,000	0.021	0.000094	0.11	0.00050
B3	31.1	164	1,450	329,000	0.021	0.000095	0.11	0.00050
B4	88.8	164	1,450	329,000	0.061	0.00027	0.11	0.00050
B5	32.4	153	1,450	329,000	0.022	0.000098	0.11	0.00047
B6	32.9	177	1,450	329,000	0.023	0.00010	0.12	0.00054
B7	90.2	165	1,450	329,000	0.062	0.00027	0.11	0.00050
B8	90.8	166	1,450	329,000	0.063	0.00028	0.11	0.00050
B9	39.1	39.1	1,450	329,000	0.027	0.00012	0.027	0.00012
B10	64.3	152	1,450	329,000	0.044	0.00020	0.10	0.00046
B11	81.8	128	1,450	329,000	0.056	0.00025	0.089	0.00039
B12	32.4	212	1,450	329,000	0.022	0.000098	0.15	0.00064
B13	71.5	38.2	1,450	329,000	0.049	0.00022	0.026	0.00012
B14	88.9	164	1,450	329,000	0.061	0.00027	0.11	0.00050
B15	70.4	140	1,450	329,000	0.049	0.00021	0.097	0.00043
B16	32.8	32.8	1,450	329,000	0.023	0.00010	0.023	0.00010
B17	199	234	1,450	329,000	0.14	0.00061	0.16	0.00071
B18	64.4	187	1,450	329,000	0.044	0.00020	0.13	0.00057
B19	31.2	31.2	1,450	329,000	0.021	0.000095	0.021	0.000095
B20	113	113	1,450	329,000	0.078	0.00034	0.078	0.00034
B21	142	33.1	1,450	329,000	0.10	0.00043	0.023	0.00010
B22	32.2	118	1,450	329,000	0.022	0.000098	0.081	0.00036
B23	31.3	64.6	1,450	329,000	0.022	0.000095	0.045	0.00020
B24	166	82.7	1,450	329,000	0.11	0.00050	0.057	0.00025
B25	32.6	32.6	1,450	329,000	0.022	0.000099	0.022	0.000099
B26	32.8	64.3	1,450	329,000	0.023	0.00010	0.044	0.00020
B27	134	36.3	1,450	329,000	0.092	0.00041	0.025	0.00011
B28	164	141	1,450	329,000	0.11	0.00050	0.097	0.00043
Dibutyl phthalate (µg/kg bw/day)								
B1	584	269	1,450	329,000	0.40	0.0018	0.19	0.00082
B2	9.45	9.45	1,450	329,000	0.0065	0.000029	0.0065	0.000029
B3	9.78	9.78	1,450	329,000	0.0067	0.000030	0.0067	0.000030
B4	584	269	1,450	329,000	0.40	0.0018	0.19	0.00082
B5	9.64	9.64	1,450	329,000	0.0066	0.000029	0.0066	0.000029

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Table 5-58. HQ Calculations for Spotted Sandpiper Based on the Dietary Dose Approach Evaluated at Every Shorebird Beach Habitat Area

Area	IR_{dose}		TRVs		HQs Based on Clam Diet		HQs Based on Worm Diet	
	Clam	Worm	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
B6	9.64	9.64	1,450	329,000	0.0066	0.000029	0.0066	0.000029
B7	584	269	1,450	329,000	0.40	0.0018	0.19	0.00082
B8	584	269	1,450	329,000	0.40	0.0018	0.19	0.00082
B9	9.55	9.55	1,450	329,000	0.0066	0.000029	0.0066	0.000029
B10	19.4	9.51	1,450	329,000	0.013	0.000059	0.0066	0.000029
B11	24.6	9.39	1,450	329,000	0.017	0.000075	0.0065	0.000029
B12	9.66	9.66	1,450	329,000	0.0067	0.000029	0.0067	0.000029
B13	19.5	158	1,450	329,000	0.013	0.000059	0.11	0.00048
B14	584	269	1,450	329,000	0.40	0.0018	0.19	0.00082
B15	293	10.4	1,450	329,000	0.20	0.00089	0.0072	0.000032
B16	9.64	164	1,450	329,000	0.0066	0.000029	0.11	0.00050
B17	302	571	1,450	329,000	0.21	0.00092	0.39	0.0017
B18	18.8	9.45	1,450	329,000	0.013	0.000057	0.0065	0.000029
B19	9.49	134	1,450	329,000	0.0065	0.000029	0.093	0.00041
B20	9.64	135	1,450	329,000	0.0066	0.000029	0.093	0.00041
B21	1519	164	1,450	329,000	1.0	0.0046	0.11	0.00050
B22	9.61	187	1,450	329,000	0.0066	0.000029	0.13	0.00057
B23	9.45	164	1,450	329,000	0.0065	0.000029	0.11	0.00050
B24	19.5	147	1,450	329,000	0.013	0.000059	0.10	0.00045
B25	9.64	158	1,450	329,000	0.0066	0.000029	0.11	0.00048
B26	10.1	108	1,450	329,000	0.0070	0.000031	0.074	0.00033
B27	9.66	205	1,450	329,000	0.0067	0.000029	0.14	0.00062
B28	93.7	54.0	1,450	329,000	0.065	0.00028	0.037	0.00016
PCB TEQ (pg/g bw/day)								
B1	ND	ND	14	140	ND	ND	ND	ND
B2	27.9	91.3	14	140	**2.0**	0.20	**6.5**	0.65
B3	73.0	573	14	140	**5.2**	0.52	**41**	**4.1**
B4	nd	nd	14	140	nd	nd	nd	nd
B5	5.62	4.46	14	140	0.40	0.040	0.32	0.032
B6	33.5	24.9	14	140	**2.4**	0.24	**1.8**	0.18
B7	nd	nd	14	140	nd	nd	nd	nd
B8	nd	nd	14	140	nd	nd	nd	nd
B9	11.2	18.0	14	140	0.80	0.080	**1.3**	0.13
B10	1.76	8.77	14	140	0.13	0.013	0.63	0.063
B11	10.7	10.7	14	140	0.76	0.076	0.77	0.077
B12	7.49	31.1	14	140	0.53	0.053	**2.2**	0.22
B13	8.86	11.9	14	140	0.63	0.063	0.85	0.085
B14	1.12	1.17	14	140	0.080	0.0080	0.084	0.0084
B15	16.1	17.0	14	140	**1.1**	0.11	**1.2**	0.12

BZTO104(e)030939

Table 5-58. HQ Calculations for Spotted Sandpiper Based on the Dietary Dose Approach Evaluated at Every Shorebird Beach Habitat Area

Area	IR_{dose}		TRVs		HQs Based on Clam Diet		HQs Based on Worm Diet	
	Clam	Worm	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
B16	15.0	393	14	140	**1.1**	0.11	**28**	**2.8**
B17	11.0	23.3	14	140	0.79	0.079	**1.7**	0.17
B18	5.70	6.03	14	140	0.41	0.041	0.43	0.043
B19	9.69	22.2	14	140	0.69	0.069	**1.6**	0.16
B20	12.9	24.2	14	140	0.92	0.092	**1.7**	0.17
B21	10.1	20.4	14	140	0.72	0.072	**1.5**	0.15
B22	84.5	855	14	140	**6.0**	0.60	**61**	**6.1**
B23	4.06	5.14	14	140	0.29	0.029	0.37	0.037
B24	10.5	19.9	14	140	0.75	0.075	**1.4**	0.14
B25	4.85	4.64	14	140	0.35	0.035	0.33	0.033
B26	14.3	8.25	14	140	1.0	0.10	0.59	0.059
B27	21.3	49.0	14	140	**1.5**	0.15	**3.5**	0.35
B28	5.01	5.59	14	140	0.36	0.036	0.40	0.040
Total PCBs (µg/kg bw/day)								
B1	12.4	12.0	290	580	0.043	0.021	0.041	0.021
B2	189	476	290	580	0.65	0.33	**1.6**	0.82
B3	437	3520	290	580	**1.5**	0.75	**12**	**6.1**
B4	23.7	42.4	290	580	0.082	0.041	0.15	0.073
B5	82.8	57.7	290	580	0.29	0.14	0.20	0.10
B6	358	353	290	580	**1.2**	0.62	**1.2**	0.61
B7	14.0	16.2	290	580	0.048	0.024	0.056	0.028
B8	13.9	16.1	290	580	0.048	0.024	0.056	0.028
B9	117	173	290	580	0.40	0.20	0.60	0.30
B10	24.8	153	290	580	0.085	0.043	0.53	0.26
B11	148	126	290	580	0.51	0.26	0.44	0.22
B12	105	387	290	580	0.36	0.18	**1.3**	0.67
B13	180	330	290	580	0.62	0.31	**1.1**	0.57
B14	12.3	11.6	290	580	0.042	0.021	0.040	0.020
B15	3110	4570	290	580	**11**	**5.4**	**16**	**7.9**
B16	132	2210	290	580	0.46	0.23	**7.6**	**3.8**
B17	211	248	290	580	0.73	0.36	0.86	0.43
B18	113	95.9	290	580	0.39	0.20	0.33	0.17
B19	105	194	290	580	0.36	0.18	0.67	0.33
B20	218	574	290	580	0.75	0.38	**2.0**	0.99
B21	282	525	290	580	0.97	0.49	**1.8**	0.90
B22	494	5040	290	580	**1.7**	0.85	**17**	**8.7**
B23	101	138	290	580	0.35	0.17	0.47	0.24
B24	452	854	290	580	**1.6**	0.78	**2.9**	**1.5**
B25	71.7	79.1	290	580	0.25	0.12	0.27	0.14
B26	192	203	290	580	0.66	0.33	0.70	0.35
B27	549	1710	290	580	**1.9**	0.95	**5.9**	**2.9**

BZTO104(e)030940

Table 5-58. HQ Calculations for Spotted Sandpiper Based on the Dietary Dose Approach Evaluated at Every Shorebird Beach Habitat Area

Area	IR_{dose}		TRVs		HQs Based on Clam Diet		HQs Based on Worm Diet	
	Clam	Worm	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
B28	59.3	86.8	290	580	0.20	0.10	0.30	0.15
Dioxin TEQ (pg/g bw/day)								
B1	0.183	0.443	14	140	0.013	0.0013	0.032	0.0032
B2	1.95	4.91	14	140	0.14	0.014	0.35	0.035
B3	2.47	10.6	14	140	0.18	0.018	0.76	0.076
B4	0.187	0.467	14	140	0.013	0.0013	0.033	0.0033
B5	1.28	4.33	14	140	0.092	0.0092	0.31	0.031
B6	2.29	5.59	14	140	0.16	0.016	0.40	0.040
B7	ND	nd	14	140	nd	nd	nd	nd
B8	nd	nd	14	140	nd	nd	nd	nd
B9	10.6	46.0	14	140	0.76	0.076	**3.3**	0.33
B10	2.72	14.6	14	140	0.19	0.019	**1.0**	0.10
B11	4.39	5.07	14	140	0.31	0.031	0.36	0.036
B12	3.52	30.9	14	140	0.25	0.025	**2.2**	0.22
B13	1.85	5.48	14	140	0.13	0.013	0.39	0.039
B14	0.203	0.584	14	140	0.014	0.0014	0.042	0.0042
B15	1.85	7.85	14	140	0.13	0.013	0.56	0.056
B16	28.0	1091	14	140	**2.0**	0.20	**78**	**7.8**
B17	45.3	153	14	140	**3.2**	0.32	**11**	**1.1**
B18	2.57	9.15	14	140	0.18	0.018	0.65	0.065
B19	0.820	7.98	14	140	0.059	0.0059	0.57	0.057
B20	3.09	6.12	14	140	0.22	0.022	0.44	0.044
B21	1.98	6.93	14	140	0.14	0.014	0.50	0.050
B22	3.21	14.7	14	140	0.23	0.023	**1.1**	0.11
B23	1.20	3.06	14	140	0.086	0.0086	0.22	0.022
B24	2.59	7.33	14	140	0.18	0.018	0.52	0.052
B25	1.39	2.21	14	140	0.10	0.010	0.16	0.016
B26	1.86	2.50	14	140	0.13	0.013	0.18	0.018
B27	4.07	6.04	14	140	0.29	0.029	0.43	0.043
B28	0.891	2.25	14	140	0.064	0.0064	0.16	0.016
Aldrin (µg/kg bw/day)								
B1	0.0729	0.0807	8.0	40	0.0091	0.0018	0.010	0.0020
B2	0.200	0.392	8.0	40	0.025	0.0050	0.049	0.0098
B3	0.261	0.276	8.0	40	0.033	0.0065	0.035	0.0069
B4	0.0716	0.0774	8.0	40	0.0090	0.0018	0.0097	0.0019
B5	0.171	0.0882	8.0	40	0.021	0.0043	0.011	0.0022
B6	0.227	0.512	8.0	40	0.028	0.0057	0.064	0.013
B7	0.118	0.202	8.0	40	0.015	0.0030	0.025	0.0051
B8	0.118	0.202	8.0	40	0.015	0.0030	0.025	0.0051
B9	0.224	0.526	8.0	40	0.028	0.0056	0.066	0.013
B10	0.0241	0.741	8.0	40	0.0030	0.00060	0.093	0.019

Table 5-58. HQ Calculations for Spotted Sandpiper Based on the Dietary Dose Approach Evaluated at Every Shorebird Beach Habitat Area

Area	IR_{dose}		TRVs		HQs Based on Clam Diet		HQs Based on Worm Diet	
	Clam	Worm	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
B11	0.262	0.501	8.0	40	0.033	0.0066	0.063	0.013
B12	0.256	0.105	8.0	40	0.032	0.0064	0.013	0.0026
B13	0.450	0.301	8.0	40	0.056	0.011	0.038	0.0075
B14	0.0708	0.0753	8.0	40	0.0089	0.0018	0.0094	0.0019
B15	0.587	0.735	8.0	40	0.073	0.015	0.092	0.018
B16	0.498	0.475	8.0	40	0.062	0.012	0.059	0.012
B17	0.672	0.677	8.0	40	0.084	0.017	0.085	0.017
B18	0.227	0.680	8.0	40	0.028	0.0057	0.085	0.017
B19	0.284	0.546	8.0	40	0.036	0.0071	0.068	0.014
B20	0.379	0.631	8.0	40	0.047	0.0095	0.079	0.016
B21	0.342	0.530	8.0	40	0.043	0.0085	0.066	0.013
B22	5.92	43.2	8.0	40	0.74	0.15	**5.4**	**1.1**
B23	0.158	0.524	8.0	40	0.020	0.0040	0.065	0.013
B24	0.342	0.544	8.0	40	0.043	0.0085	0.068	0.014
B25	0.204	0.498	8.0	40	0.025	0.0051	0.062	0.012
B26	0.433	0.442	8.0	40	0.054	0.011	0.055	0.011
B27	0.413	0.596	8.0	40	0.052	0.010	0.075	0.015
B28	0.150	0.478	8.0	40	0.019	0.0038	0.060	0.012
Sum DDD (µg/kg bw/day)								
B1	4.23	4.45	180	900	0.024	0.0047	0.025	0.0049
B2	11.1	18.1	180	900	0.062	0.012	0.10	0.020
B3	13.6	16.6	180	900	0.076	0.015	0.092	0.018
B4	1.25	1.01	180	900	0.0070	0.0014	0.0056	0.0011
B5	12.3	13.1	180	900	0.06814	0.014	0.073	0.015
B6	12.5	27.0	180	900	0.070	0.014	0.15	0.030
B7	1.34	1.11	180	900	0.0075	0.0015	0.0062	0.0012
B8	2.89	2.90	180	900	0.016	0.0032	0.016	0.0032
B9	7.15	13.0	180	900	0.040	0.0079	0.072	0.014
B10	1.14	28.4	180	900	0.0063	0.0013	0.16	0.032
B11	19.0	25.1	180	900	0.11	0.021	0.14	0.028
B12	20.5	92.0	180	900	0.11	0.023	0.51	0.10
B13	13.2	31.8	180	900	0.073	0.015	0.18	0.035
B14	1.36	1.13	180	900	0.0075	0.0015	0.0063	0.0013
B15	7.50	24.4	180	900	0.042	0.0083	0.14	0.027
B16	233	1475	180	900	**1.3**	0.26	**8.2**	**1.6**
B17	284	1048	180	900	**1.6**	0.32	**5.8**	**1.2**
B18	5.20	13.3	180	900	0.029	0.0058	0.074	0.015
B19	17.0	34.6	180	900	0.094	0.019	0.19	0.038
B20	12.0	25.3	180	900	0.067	0.013	0.14	0.028

Table 5-58. HQ Calculations for Spotted Sandpiper Based on the Dietary Dose Approach Evaluated at Every Shorebird Beach Habitat Area

Area	IR_{dose}		TRVs		HQs Based on Clam Diet		HQs Based on Worm Diet	
	Clam	Worm	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
B21	10.3	21.7	180	900	0.057	0.011	0.12	0.024
B22	34.9	153	180	900	0.19	0.039	0.85	0.17
B23	3.18	10.3	180	900	0.018	0.0035	0.0.57	0.011
B24	7.93	12.5	180	900	0.044	0.0088	0.0.070	0.014
B25	3.49	6.99	180	900	0.020	0.0039	0.039	0.0078
B26	5.62	15.0	180	900	0.031	0.0062	0.0.83	0.017
B27	6.92	14.2	180	900	0.038	0.0077	0.079	0.016
B28	2.52	5.59	180	900	0.014	0.0028	0.031	0.0062
Sum DDE (µg/kg bw/day)								
B1	2.21	2.34	64	320	0.035	0.0069	0.037	0.0073
B2	14.5	29.5	64	320	0.23	0.045	0.46	0.092
B3	17.0	21.0	64	320	0.27	0.053	0.33	0.066
B4	2.05	1.84	64	320	0.032	0.0064	0.029	0.0058
B5	12.3	13.7	64	320	0.19	0.038	0.21	0.043
B6	14.4	24.5	64	320	0.22	0.045	0.38	0.077
B7	2.15	2.15	64	320	0.034	0.0067	0.034	0.0067
B8	2.73	3.90	64	320	0.043	0.0085	0.061	0.012
B9	10.0	15.0	64	320	0.16	0.031	0.23	0.047
B10	0.861	32.0	64	320	0.013	0.0027	0.50	0.10
B11	16.9	21.4	64	320	0.26	0.053	0.33	0.067
B12	17.9	53.7	64	320	0.28	0.056	0.84	0.17
B13	18.0	19.5	64	320	0.28	0.056	0.30	0.061
B14	2.39	2.86	64	320	0.037	0.0075	0.045	0.0089
B15	11.0	32.9	64	320	0.17	0.035	0.51	0.10
B16	77.1	223	64	320	**1.2**	0.24	**3.5**	0.70
B17	129	102	64	320	**2.0**	0.40	**1.6**	0.32
B18	9.98	23.8	64	320	0.16	0.031	0.37	0.074
B19	17.2	30.1	64	320	0.27	0.054	0.47	0.094
B20	17.1	37.1	64	320	0.27	0.054	0.58	0.12
B21	16.3	27.5	64	320	0.25	0.051	0.43	0.086
B22	78.2	179	64	320	**1.2**	0.24	**2.8**	0.56
B23	7.33	16.6	64	320	0.11	0.023	0.26	0.052
B24	12.0	14.3	64	320	0.19	0.038	0.22	0.045
B25	9.34	16.5	64	320	0.15	0.029	0.26	0.051
B26	15.0	18.1	64	320	0.23	0.047	0.28	0.057
B27	11.1	13.6	64	320	0.17	0.035	0.21	0.043
B28	5.73	11.3	64	320	0.090	0.018	0.18	0.035
Sum DDT (µg/kg bw/day)								
B1	7.36	20.9	30	150	0.25	0.049	0.70	0.14

Table 5-58. HQ Calculations for Spotted Sandpiper Based on the Dietary Dose Approach Evaluated at Every Shorebird Beach Habitat Area

Area	IR_{dose}		TRVs		HQs Based on Clam Diet		HQs Based on Worm Diet	
	Clam	Worm	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
B2	1.89	0.232	30	150	0.063	0.013	0.0077	0.0015
B3	2.81	0.842	30	150	0.094	0.019	0.028	0.0056
B4	0.722	1.05	30	150	0.024	0.0048	0.035	0.0070
B5	2.05	1.68	30	150	0.068	0.014	0.056	0.011
B6	3.98	0.750	30	150	0.13	0.027	0.025	0.0050
B7	0.643	0.809	30	150	0.021	0.0043	0.027	0.0054
B8	0.698	0.974	30	150	0.023	0.0047	0.032	0.0065
B9	2.47	0.179	30	150	0.082	0.016	0.0060	0.0012
B10	0.166	0.374	30	150	0.0055	0.0011	0.012	0.0025
B11	4.46	0.406	30	150	0.15	0.030	0.014	0.0027
B12	4.85	2.42	30	150	0.16	0.032	0.081	0.016
B13	1.87	3.33	30	150	0.062	0.012	0.11	0.022
B14	0.997	1.87	30	150	0.033	0.0066	0.062	0.012
B15	4.85	0.593	30	150	0.16	0.032	0.020	0.0040
B16	56.2	44.3	30	150	**1.9**	0.37	**1.5**	0.30
B17	143	56.4	30	150	**4.8**	0.95	**1.9**	0.38
B18	1.52	0.179	30	150	0.051	0.010	0.0060	0.0012
B19	4.56	0.566	30	150	0.15	0.030	0.019	0.0038
B20	2.44	0.361	30	150	0.081	0.016	0.012	0.0024
B21	4.75	2.05	30	150	0.16	0.032	0.068	0.014
B22	1.93	0.139	30	150	0.064	0.013	0.0046	0.00093
B23	0.964	0.195	30	150	0.032	0.0064	0.0065	0.0013
B24	2.14	0.340	30	150	0.071	0.014	0.011	0.0023
B25	1.48	0.160	30	150	0.049	0.0098	0.0053	0.0011
B26	1.44	1.91	30	150	0.048	0.0096	0.064	0.013
B27	3.07	0.703	30	150	0.10	0.020	0.023	0.0047
B28	0.923	0.0804	30	150	0.031	0.0062	0.0027	0.00054
Total DDTs (µg/kg bw/day)								
B1	12.2	23.3	180	1,800	0.068	0.0068	0.13	0.013
B2	27.5	47.9	180	1,800	0.15	0.015	0.27	0.027
B3	33.4	38.6	180	1,800	0.19	0.019	0.21	0.021
B4	2.46	2.36	180	1,800	0.014	0.0014	0.013	0.0013
B5	26.7	28.6	180	1,800	0.15	0.015	0.16	0.016
B6	30.9	52.0	180	1,800	0.17	0.017	0.29	0.029
B7	2.34	2.10	180	1,800	0.013	0.0013	0.012	0.0012
B8	5.14	8.11	180	1,800	0.029	0.0029	0.045	0.0045
B9	19.7	28.2	180	1,800	0.11	0.011	0.16	0.016
B10	2.17	60.8	180	1,800	0.012	0.0012	0.34	0.034
B11	40.4	46.9	180	1,800	0.22	0.022	0.26	0.026

Table 5-58. HQ Calculations for Spotted Sandpiper Based on the Dietary Dose Approach Evaluated at Every Shorebird Beach Habitat Area

Area	IR_{dose}		TRVs		HQs Based on Clam Diet		HQs Based on Worm Diet	
	Clam	Worm	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
B12	43.3	148	180	1,800	0.24	0.024	0.82	0.082
B13	33.1	54.6	180	1,800	0.18	0.018	0.30	0.030
B14	3.14	3.82	180	1,800	0.017	0.0017	0.021	0.0021
B15	22.2	57.9	180	1,800	0.12	0.012	0.32	0.032
B16	367	1,750	180	1,800	**2.0**	0.20	**9.7**	0.97
B17	556	1,210	180	1,800	**3.1**	0.31	**6.7**	0.67
B18	16.7	37.4	180	1,800	0.093	0.0093	0.21	0.021
B19	38.8	65.3	180	1,800	0.22	0.022	0.36	0.036
B20	31.5	62.8	180	1,800	0.17	0.017	0.35	0.035
B21	31.3	51.2	180	1,800	0.17	0.017	0.28	0.028
B22	115	332	180	1,800	0.64	0.064	**1.8**	0.18
B23	11.5	27.0	180	1,800	0.064	0.0064	0.15	0.015
B24	22.1	27.0	180	1,800	0.12	0.012	0.15	0.015
B25	14.3	23.6	180	1,800	0.079	0.0079	0.13	0.013
B26	22.1	34.9	180	1,800	0.12	0.012	0.19	0.019
B27	21.2	28.5	180	1,800	0.12	0.012	0.16	0.016
B28	9.15	17.0	180	1,800	0.051	0.0051	0.094	0.0094

BEHP – bis(2-ethylhexyl) phthalate
bw – body weight
HQ – hazard quotient
LOAEL – low-observed-adverse-effect level
NA – no LOAEL TRV available
NE – not evaluated; no TRV available
ND – no sediment data; IR and HQs not evaluated
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
PAH – polycyclic aromatic hydrocarbon

Table 5-59. HQ Calculations for Hooded Merganser Based on the Dietary Dose Approach

Round 2 COPC	Unit (bw/day)	IR_{diet}	TRVs		HQ		Contribution to Risk (%)	
			NOAEL	LOAEL	NOAEL	LOAEL	Tissue	Sediment
Copper	mg/kg	1.69	47	62	0.036	0.027	91.3%	8.7%
Lead	mg/kg	0.886	2	20	0.44	0.044	84.6%	15.4%
Mercury	mg/kg	0.0140	0.01	0.05	**1.4**	0.28	98.1%	1.9%
Benzo(a)pyrene	µg/kg	35.0	280	1,400	0.13	0.025	76.2%	23.8%
BEHP	µg/kg	3390	1,450	329,000	**2.3**	0.010	99.7%	0.33%
PCB TEQ	pg/g	28.7	14	140	**2.1**	0.21	97.2%	2.8%
Total PCBs	µg/kg	376	290	580	**1.3**	0.65	99.7%	0.35%
Dioxin TEQ	pg/g	62.2	14	140	**4.4**	0.44	89.0%	11.0%
Sum DDE	µg/kg	83.9	64	320	**1.3**	0.26	99.9%	0.15%
Sum DDT	µg/kg	63.7	30	150	**2.1**	0.42	98.3%	1.7%
Total DDTs	µg/kg	355	180	1,800	**2.0**	0.20	99.6%	0.40%

BEHP – bis(2-ethylhexyl) phthalate
bw – body weight
COPC – chemical of potential concern
HQ – hazard quotient
IR – ingestion rage
LOAEL – low-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
TRV – toxicity reference value

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Table 5-60. HQ Calculations for Bald Eagle Based on the Dietary Dose Approach

Round 2 COPC	Unit (bw/day)	IR_{diet}	TRVs		HQs		Contribution to Risk (%)	
			NOAEL	LOAEL	NOAEL	LOAEL	Tissue	Sediment
Lead	mg/kg	0.282	2	20	0.14	0.014	85.2%	14.8%
Mercury	mg/kg	0.00959	0.01	0.05	0.96	0.19	99.2%	0.85%
BEHP	µg/kg	819	1450	329,000	0.56	0.0025	99.6%	0.42%
PCB TEQ	pg/g	4.11	14	140	0.29	0.029	93.9%	6.1%
Total PCBs	µg/kg	443	290	580	**1.5**	0.76	99.9%	0.090%
Dioxin TEQ	pg/g	4.89	14	140	0.35	0.035	57.1%	42.9%
Sum DDE	µg/kg	22.6	64	320	0.35	0.071	99.8%	0.17%
Sum DDT	µg/kg	20.6	30	150	0.69	0.14	98.4%	1.6%

BEHP – bis(2-ethylhexyl) phthalate
bw – body weight
COPC – chemical of potential concern
HQ – hazard quotient
IR – ingestion rage
LOAEL – low-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
TRV – toxicity reference value

Table 5-61. HQ Calculations for Osprey Based on the Dietary Dose Approach

Round 2 COPC	Unit (bw/day)	IR_{diet}	TRVs		HQs		Contribution to Risk (%)	
			NOAEL	LOAEL	NOAEL	LOAEL	Tissue	Sediment
Lead	mg/kg	0.105	2	20	0.052	0.0052	30.2%	69.8%
Mercury	mg/kg	0.0213	0.01	0.05	**2.1**	0.43	99.3%	0.67%
Benzo(a)pyrene	µg/kg	7.87	280	1,400	0.028	0.0056	43.2%	56.8%
BEHP	µg/kg	2,384	1,450	329,000	**1.6**	0.0072	99.7%	0.25%
PCB TEQ	pg/g	6.59	14	140	0.47	0.047	93.4%	6.6%
Total PCBs	µg/kg	383	290	580	**1.3**	0.66	99.8%	0.18%
Dioxin TEQ	pg/g	6.52	14	140	0.47	0.047	43.8%	56.2%
Sum DDE	µg/kg	37.4	64	320	0.58	0.12	99.8%	0.18%
Sum DDT	µg/kg	51.9	30	150	**1.7**	0.35	98.9%	1.1%

BEHP – bis(2-ethylhexyl) phthalate
bw – body weight
COPC – chemical of potential concern
HQ – hazard quotient
IR – ingestion rage
LOAEL – low-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
TRV – toxicity reference value

Table 5-62. HQ Calculations for Mink Based on the Dietary Dose Approach

Round 2 COPC	Unit (bw/day)	IR_{diet}	TRVs		HQs		Contribution to Risk (%)	
			NOAEL	LOAEL	NOAEL	LOAEL	Tissue	Sediment
Antimony	mg/kg	0.00729	1,489	na	0.0000049	ne	6.1%	93.9%
Copper	mg/kg	0.791	18	26	0.044	0.030	64.4%	35.6%
Lead	mg/kg	0.587	11.3	90.3	0.052	0.0065	55.3%	44.7%
Mercury	mg/kg	0.0121	0.02	0.07	0.77	0.16	95.8%	4.2%
Selenium	mg/kg	0.0512	0.05	0	0.93	0.64	80.9%	19.1%
Total PAHs	µg/kg	351.8	2,000	10,000	0.18	0.035	26.6%	73.4%
PCB TEQ	pg/g	3.50	0.44	2.24	**8.0**	**1.6**	96.4%	3.6%
Total PCBs	µg/kg	306	7.4	37	**41**	**8.3**	99.2%	0.8%
Dioxin TEQ	pg/g	7.91	0.44	2.24	**18**	**3.5**	21.9%	78.1%
Total DDTs	µg/kg	75.6	260	1,300	0.29	0.058	96.4%	3.6%

bw – body weight
na – no TRV available
ne – not evaluated; no TRV available
COPC – chemical of potential concern
HQ – hazard quotient
IR – ingestion rate
LOAEL – low-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
TRV – toxicity reference value

Table 5-63. HQ Calculations for River Otter Based on the Dietary Dose Approach

Round 2 COPC	Unit (bw/day)	IR_{diet}	TRVs		HQs		Contribution to Risk (%)	
			NOAEL	LOAEL	NOAEL	LOAEL	Tissue	Sediment
Total PAHs	µg/kg	104	2,000	10,000	0.052	0.010	61.7%	38.3%
PCB TEQ	pg/g	2.64	0.44	2.24	**6.0**	**1.2**	99.3%	0.74%
Total PCBs	µg/kg	253	7.4	37	**34**	**6.8**	99.8%	0.15%
Dioxin TEQ	pg/g	2.06	0.44	2.24	**4.7**	0.92	53.6%	46.4%
Total DDTs	µg/kg	62.6	260	1,300	0.24	0.048	99.3%	0.68%

bw – body weight
COPC – chemical of potential concern
HQ – hazard quotient
IR – ingestion rage
LOAEL – low-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
TRV – toxicity reference value

Table 5-64. HQ Calculations for Osprey and Bald Eagle Based on the Bird Egg Approach

Receptor	Round 2 COPC	Units (ww)	EPC_{egg}	TRVs		HQs	
				NOAEL	LOAEL	NOAEL	LOAEL
Bald eagle	dioxin TEQ	ng/kg	233	2.3	31.98	**100**	**7.3**
	PCB TEQ	ng/kg	332	2.3	31.98	**140**	**10**
	total PCBs	µg/kg	40,545	3,000	4,500	**14**	**9.0**
	mercury	mg/kg	0.23775	0.5	0.74	0.48	0.32
	sum DDE	µg/kg	16,339	1,300	3,500	**13**	**4.7**
Osprey	dioxin TEQ	ng/kg	136	2.3	31.98	**59**	**4.3**
	PCB TEQ	ng/kg	293	2.3	31.98	**130**	**9.2**
	total PCBs	µg/kg	19,998	3,000	4,500	**6.7**	**4.4**
	mercury	mg/kg	0.30207	0.5	0.74	0.60	0.41
	sum DDE	µg/kg	15,468	1,300	3,500	**12**	**4.4**

COPC – chemical of potential concern
EPC – exposure point concentration
HQ – hazard quotient
LOAEL – low-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
TRV – toxicity reference value
ww – wet weight

Table 5-65. Dioxin TEQ and PCB TEQ HQ Calculations for Mink and River Otter

Receptor	Round 2 COPC	Unit (bw/day)	IR_{diet}	TRVs		HQs	
				NOAEL	LOAEL	NOAEL	LOAEL
Mink	dioxin TEQ (1998)	pg/g	7.91	0.44	2.24	**18**	**3.5**
	dioxin TEQ (2006)	pg/g	6.65			**15**	**3.0**
	PCB TEQ (1998)	pg/g	3.50	0.44	2.24	**8.0**	**1.6**
	PCB TEQ (2006)	pg/g	1.54			**3.5**	0.69
River otter	dioxin TEQ (1998)	pg/g	2.06	0.44	2.24	**4.7**	0.92
	dioxin TEQ (2006)	pg/g	1.70			**3.9**	0.76
	PCB TEQ (1998)	pg/g	2.64	0.44	2.24	**6.0**	**1.2**
	PCB TEQ (2006)	pg/g	1.05			**2.4**	0.47

COPC – chemical of potential concern
HQ – hazard quotient
IR – ingestion rate
LOAEL – low-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
TRV – toxicity reference value

Table 5-66. Results of Uncertainty Analysis of Varying Dietary Prey Portions on HQ Results

Species	Change in HQ[a]	Prey Item with Maximum EPC	F_{prey} of Prey Item with Maximum EPC	Approximate Change F_{prey} Required for Change in HQs	Likelihood of Occurrence
Lead					
Bald eagle	NOAEL HQ > 1.0	peamouth	5%	peamouth portion would have to be ≥ 44%	low; literature does not support high consumption of peamouth by bald eagle
Hooded merganser	NOAEL HQ > 1.0	peamouth	5%	peamouth portion would have to be ≥ 14%	low; literature does not support consumption of fish larger than 2 inches; peamouth do not represent a small fish prey item for hooded merganser
Mercury					
Bald eagle	NOAEL HQ > 1.0	northern pikeminnow	5%	northern pikeminnow portion would have to be ≥ 6%	high; literature does not support high consumption of pikeminnow by bald eagle, but slight increase is likely
Osprey	NOAEL HQ magnitude doubled and LOAEL HQ > 1.0	northern pikeminnow	7%	northern pikeminnow portion would have to be ≥ 38%; (and ≥ 49% for LOAEL HQ >1)	low; literature does not support high consumption of northern pikeminnow by osprey
Mink	NOAEL HQ > 1.0	northern pikeminnow	5%	northern pikeminnow portion would have to be ≥ 12%	high; uncertainty in mink diet composition because as an opportunistic feeder, may prey on northern pikeminnow more frequently than assumed (5%)
Selenium					
Mink	NOAEL and LOAEL HQ > 1.0	peamouth	5%	peamouth portion would have to be ≥ 13% (and ≥ 86% for LOAEL > 1.0)	low; literature does not support high consumption of peamouth by mink
BEHP					
Bald eagle	NOAEL HQ > 1.0	northern pikeminnow	5%	northern pikeminnow portion would have to be ≥ 13%	low; literature does not support high consumption of northern pikeminnow by bald eagle
Osprey	NOAEL HQ magnitude doubled	northern pikeminnow and smallmouth bass	7% and 2%, respectively	both smallmouth bass and northern pikeminnow portion would have to be ≥ 26%	low; literature does not support high consumption of northern pikeminnow or smallmouth bass by osprey
Total PCBs					
Bald eagle	LOAEL HQ > 1.0	carp	45%	carp portion would have to be ≥ 67%	moderate; literature supports carp as primary prey item of carp by bald eagle; however, unknown whether sufficient prey to make up 2/3 of bald eagle diet based on carp

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Table 5-66. Results of Uncertainty Analysis of Varying Dietary Prey Portions on HQ Results

Species	Change in HQ[a]	Prey Item with Maximum EPC	F_{prey} of Prey Item with Maximum EPC	Approximate Change F_{prey} Required for Change in HQs	Likelihood of Occurrence
Osprey	NOAEL HQ magnitude doubled; LOAEL HQ > 1.0	carp	6%	carp portion would have to be ≥ 43% (and ≥ 25% for LOAEL HQ >1)	low; literature does not support high consumption of carp by osprey
Mink	NOAEL HQ and LOAEL HQ magnitude doubled	carp	15%	carp portion would have to be ≥ 47%	moderate; literature does not support high consumption of carp by mink; however, uncertainty in mink diet composition because as an opportunistic feeder, may prey on carp more frequently than assumed (15%)
River otter	NOAEL HQ and LOAEL HQ magnitude doubled	carp	26%	carp portion would have to be ≥ 73%	moderate; literature supports carp as primary prey item of carp by river otter; however, unknown whether sufficient prey to make up 3/4 of river otter diet based on carp
Dioxin TEQ					
Bald eagle	NOAEL HQ > 1.0	peamouth (sculpin used as surrogate)	5%	peamouth portion would have to be ≥ 46%	low; literature does not support high consumption of peamouth by bald eagle and use of sculpin as surrogate does not represent realistic prey item for bald eagles
River otter	LOAEL HQ > 1.0	crayfish	10%	crayfish portion would have to be ≥ 19% of diet	moderate; literature suggests approximately 10% of otter diet is decapods (crabs); however, unknown whether sufficient prey to make up 19% of river otter diet based on crayfish
Sum DDE					
Osprey	NOAEL HQ > 1.0	northern pikeminnow	7%	northern pikeminnow portion would have to be ≥ 57%	low; literature does not support high consumption of northern pikeminnow by osprey
Sum DDT					
Bald Eagle	NOAEL HQ > 1.0	largescale sucker	45%	largescale sucker portion would have to be ≥ 85%	moderate; literature supports largescale sucker as primary prey item of bald eagle; however, unknown whether sufficient prey to make up 85% of bald eagle diet

[a] This column reflects the difference in HQs based on 100% ingestion of the prey item with the highest EPC from HQs presented in Tables 5-69 through 5-73.

EPC – exposure point concentration

HQ – hazard quotient

TEQ – toxic equivalent

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Table 5-67. Average Percent of Total Risk for Wildlife Receptors from Prey tissue, Sediment, and Surface Water Pathways

Wildlife Receptor	Average Percentage of Total Risk		
	Prey Tissue	Sediment	Surface Water
Hooded merganser	87.6%	12.4%	0.017%
Bald eagle	87.2%	12.8%	0.018%
Osprey	84.1%	15.9%	0.018%
Mink	73.5%	26.5%	0.033%
River otter	84.3%	15.6%	0.062%

Table 5-68. Comparison of Dietary Dose HQ Calculations for Hooded Merganser, Osprey, and Bald Eagle Using Different SUF Assumptions

Receptor	Round 2 COPC	HQs (100% SUF)		HQs (50% SUF)	
		NOAEL	LOAEL	NOAEL	LOAEL
Hooded merganser	copper	0.036	0.027	0.018	0.014
	lead	0.44	0.044	0.22	0.022
	mercury	**1.4**	0.28	0.70	0.14
	benzo(a)pyrene	0.13	0.025	0.065	0.013
	BEHP	**2.3**	0.010	**1.2**	0.0050
	PCB TEQ	**2.1**	0.21	**1.1**	0.11
	total PCBs	**1.3**	0.65	0.65	0.33
	dioxin TEQ	**4.4**	0.44	**2.2**	0.22
	sum DDE	**1.3**	0.26	0.65	0.13
	sum DDT	**2.1**	0.42	**1.1**	0.21
	total DDTs	**2.0**	0.20	1.0	0.10
Bald eagle	dioxin TEQ	0.35	0.035	0.18	0.018
	PCB TEQ	0.29	0.029	0.15	0.015
	total PCBs	**1.5**	0.76	0.75	0.38
	lead	0.14	0.014	0.070	0.0070
	mercury	0.96	0.19	0.48	0.095
	sum DDE	0.35	0.071	0.18	0.036
	sum DDT	0.69	0.14	0.35	0.070
	BEHP	0.56	0.0025	0.28	0.0013
Osprey	dioxin TEQ	0.47	0.047	0.24	0.024
	PCB TEQ	0.47	0.047	0.24	0.024
	total PCBs	**1.3**	0.66	0.65	0.33
	lead	0.052	0.0052	0.026	0.0026
	mercury	**2.1**	0.43	**1.1**	0.22
	benzo(a)pyrene	0.028	0.0056	0.014	0.0028
	sum DDE	0.58	0.12	0.29	0.060
	sum DDT	**1.7**	0.35	0.85	0.18
	BEHP	**1.6**	0.0072	0.80	0.0036

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – low-observed-adverse-effect level
ne – not evaluated; no TRV available
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SUF – site use factor
TEQ – toxic equivalent
Bold identifies HQs > 1.0.

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Table 5-69. Comparison of HQs for Spotted Sandpiper Based on UCL EPCs and Mean EPCs Across All Beach Locations

Round 2 COPC	Clam Diet				Worm Diet			
	HQs Based on UCL EPCs		HQs Based on Mean EPCs		HQs Based on UCL EPCs		HQs Based on Mean EPCs	
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Chromium	**1.7**	0.32	**1.5**	0.30	**1.5**	0.30	**1.3**	0.27
Lead	0.89	0.089	0.44	0.044	**1.1**	0.11	0.58	0.058
Mercury	**1.4**	0.29	**1.2**	0.24	1.0	0.20	0.77	0.15
Benzo(a)pyrene	0.87	0.17	0.12	0.025	0.50	0.099	0.30	0.061
PCB TEQ	**2.7**	0.27	**1.3**	0.13	**31**	**3.1**	**5.7**	0.57
Total PCBs	**2.6**	**1.3**	1.0	0.51	**11**	**5.6**	**2.7**	**1.3**
Dioxin TEQ	0.95	0.095	0.40	0.040	**30**	**3.0**	**3.7**	0.37
Aldrin	0.16	0.033	0.063	0.013	**2.0**	0.41	0.25	0.050
Sum DDD	0.85	0.17	0.20	0.039	**4.0**	0.79	0.64	0.13
Sum DDE	0.71	0.14	0.37	0.074	**1.2**	0.25	0.59	0.12
Sum DDT	**2.3**	0.45	0.45	0.089	0.92	0.18	0.14	0.028
Total DDTs	0.94	0.094	0.40	0.040	**4.7**	0.47	0.87	0.087

COPC – chemical of potential concern
EPC – exposure point concentration
HQ – hazard quotient
LOAEL – low-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
UCL – upper confidence limit
Bold identifies HQs > 1.0.

Table 5-70. Comparison of HQs for Study-Area-Wide Bird Receptors Based on UCL EPCs and Mean EPCs

Round 2 COPC	Hooded Merganser				Bald Eagle				Osprey			
	HQs Based on UCL EPCs		HQs Based on Mean EPCs		HQs Based on UCL EPCs		HQs Based on Mean EPCs		HQs Based on UCL EPCs		HQs Based on Mean EPCs	
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Mercury	**1.4**	0.28	**1.2**	0.25	0.96	0.19	0.78	0.16	**2.1**	0.43	**1.7**	0.34
BEHP	**2.3**	0.010	0.27	0.0012	0.56	0.0025	0.069	0.00030	**1.6**	0.0072	0.20	0.00086
PCB TEQ	**2.1**	0.21	0.74	0.074	0.29	0.029	0.20	0.020	0.47	0.047	0.34	0.034
Total PCBs	**1.3**	0.65	0.59	0.30	**1.5**	0.76	0.54	0.27	**1.3**	0.66	0.69	0.35
Dioxin TEQ	**4.4**	0.44	0.54	0.054	0.35	0.035	0.087	0.0087	0.47	0.047	0.14	0.014
Sum DDE	**1.3**	0.26	0.29	0.058	0.35	0.071	0.26	0.051	0.58	0.12	0.43	0.085
Sum DDT	**2.1**	0.42	**1.1**	0.22	0.69	0.14	0.16	0.032	**1.7**	0.35	0.46	0.091
Total DDTs	**2.0**	0.20	0.34	0.034	NC	NC	NC	NC	NC	NC	NC	NC

COPC – chemical of potential concern
EPC – exposure point concentration
HQ – hazard quotient
LOAEL – low-observed-adverse-effect level
NC – not calculated; no LOAEL or NOAEL TRV recommended by LWG
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
UCL – upper confidence limit
Bold identifies HQs > 1.0.

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Table 5-71. Comparison of HQs for Study-Area-Wide Mammal Receptors Based on UCL EPCs and Mean EPCs

Round 2 COPC	Mink				River Otter			
	HQs Based on UCL EPCs		HQs Based on Mean EPCs		HQs Based on UCL EPCs		HQs Based on Mean EPCs	
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
PCB TEQ	**8.0**	**1.6**	**4.1**	0.81	**6.0**	**1.2**	**2.7**	0.53
Total PCBs	**41**	**8.3**	**17**	**3.4**	**34**	**6.8**	**13**	**2.5**
Dioxin TEQ	**18**	**3.5**	**3.0**	0.58	**4.7**	0.92	**1.3**	0.25

COPC – chemical of potential concern
EPC – exposure point concentration
HQ – hazard quotient
LOAEL – low-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
UCL – upper confidence limit
Bold identifies HQs > 1.0.

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Table 5-72. Comparison of Bird Egg HQ Calculations for Osprey and Bald Eagle Using Different SUF Assumptions

Receptor	Round 2 COPC	HQs (100% SUF)		HQs (50% SUF)	
		NOAEL	LOAEL	NOAEL	LOAEL
Bald eagle	mercury	0.48	0.32	0.24	0.16
	PCB TEQ (birds)	**140**	**10**	**72**	**5.0**
	total PCBs	**14**	**9.0**	**7.0**	**4.5**
	dioxin TEQ (birds)	**100**	**7.3**	**51**	**3.7**
	sum DDE	**13**	**4.7**	**6.5**	**2.4**
Osprey	mercury	0.60	0.41	0.30	0.21
	PCB TEQ (birds)	**130**	**9.2**	**64**	**4.6**
	total PCBs	**6.7**	**4.4**	**3.4**	**2.2**
	dioxin TEQ (birds)	**59**	**4.3**	**30**	**2.2**
	sum DDE	**12**	**4.4**	**6.0**	**2.2**

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – low-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SUF – site use factor
TEQ – toxic equivalent
Bold identifies HQs > 1.0.

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Table 5-73. Summary of Uncertainties for PCBs (PCB TEQs and Total PCBs)

Receptor	NOAEL HQ	LOAEL HQ	Effects Uncertainty	Exposure Uncertainty	Exposure Scale Analysis
Total PCBs					
Spotted sandpiper	0.042 – **11** (clam)	0.021 – **5.4** (clam)	moderate: TRV is based on chicken and may overestimate risks to non-domestic bird species	high: beach evaluation assumes 100% site use of individual beach areas for breeding; shorebird areas with LOAEL HQs > 1.0 are all high habitat except B3 (it is not expected that shorebirds would forage and breed here); FWM used to estimate tissues at five beach locations; diet based on 100% ingestion of worms may overestimate risks to shorebirds (ingestion of clams predicts risks at only one beach area).	On a beach-by-beach assessment of risk, clam ingestion results in LOAEL HQs <1.0 at B15 only; worm ingestion results in LOAEL HQ > 1.0 at B3, B15, B16, B22, B24, and B27 only.
	0.040 – **17** (worm)	0.020 – **8.7** (worm)			
Hooded merganser	**1.3**	0.65		low: changing dietary portions does not result in significantly higher HQs; reducing SUF to 0.5 would result in NOAEL HQ < 1.0	Reducing exposure scale to less than Study Area-wide could result in LOAEL HQ > 1.0 and < 2.0 in localized areas.
Osprey (dietary dose)	**1.3**	0.66		low: higher ingestion of carp would result in increased magnitude of NOAEL HQ and LOAEL HQ > 1.0, but likelihood of carp ingestion is low; reducing SUF to 0.5 would result in NOAEL HQ < 1.0	Reducing exposure scale to less than Study Area-wide would not significantly increase the magnitude of HQs.
Bald eagle (dietary dose)	**1.5**	0.76		moderate to high: modeled ingestion of birds and fish results in LOAEL HQ > 1.0; and higher ingestion of carp would result in LOAEL HQ > 1.0; reducing SUF to 0.5 would result in NOAEL HQ < 1.0	
Osprey (modeled egg)	**6.7**	**4.4**	moderate: TRV is based on field collected bald eagle eggs in numerous locations	moderate to high: bird egg model approach assumes literature based BMF, which may include chemical exposure from outside the Study Area (in areas where osprey winter) and overestimate exposure; reducing SUF to 0.5 would not significantly reduce HQs (all HQs still > 1.0)	Not evaluated.
Bald eagle (modeled egg)	**14**	**9.0**			
Mink	**41**	**8.3**	moderate: TRV is based on field collected fish with other contaminants present (i.e., dioxins) and may overestimate toxicity	moderate: higher ingestion of carp would result in increased magnitude of HQs	Reducing exposure scale to less than Study Area-wide would not significantly increase the magnitude of HQs.
River otter	**34**	**6.8**			

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Table 5-73. Summary of Uncertainties for PCBs (PCB TEQs and Total PCBs)

Receptor	NOAEL HQ	LOAEL HQ	Effects Uncertainty	Exposure Uncertainty	Exposure Scale Analysis
PCB TEQ					
Spotted sandpiper	0.080 – **6.0** (clam)	0.008 – 0.06 (clam)	high to moderate: TRV is based on weekly injection study; limited toxicity data available; TEFs for birds are moderately uncertain and may over or underestimate risks of dioxin-like PCBs for birds	high: beach evaluation assumes 100% site use of individual beach areas for breeding; shorebird areas with LOAEL HQs > 1.0 are all high habitat; FWM used to estimate tissues at five beach locations; diet based on 100% ingestion of worms may overestimate risks to shorebirds (ingestion of clams predicts no risks)	On a beach-by-beach assessment of risk, clam ingestion results in LOAEL HQs < 1.0 at all beaches; worm ingestion results in LOAEL HQ > 1.0 at B3, B16 and B22 only.
	0.084 – **61** (worm)	0.0084 – **6.1** (worm)			
Hooded merganser	**2.1**	0.21		low: changing dietary portions does not result in significantly higher HQs; reducing SUF to 0.5 would not significantly reduce HQs	Reducing exposure scale to less than Study Area-wide could result in NOAEL HQs > 1.0 in localized areas, but all LOAEL HQs would be < 1.0.
Osprey (diet dose)	0.47	0.047		low: changing dietary portions does not result in significantly higher HQs	
Bald eagle (diet dose)	0.29	0.029		moderate: modeled ingestion of birds and fish would result in a NOAEL HQ > 1.0 but a LOAEL HQ < 1.0	
Osprey (modeled egg)	**130**	**9.2**	moderate: TRV is based on field collected bald eagle eggs in Columbia River; TEFs for birds are moderately uncertain; may over or underestimate risks of dioxin-like PCBs for birds	moderate to high: bird egg model approach assumes literature based BMF, which may include chemical exposure from outside the Study Area (in areas where osprey winter) and overestimate exposure; reducing SUF to 0.5 would not significantly reduce HQs (all HQs still > 1.0)	Not evaluated.
Bald eagle (modeled egg)	**140**	**10**			
Mink	**8.0**	**1.6**	moderate: TRV is based on field collected fish with other contaminants present (i.e., dioxins, PCBs)	low to moderate: changing dietary portions does not result in significantly higher HQs; using updated 2006 TEFs for mammals to calculated TEQs results in LOAEL HQs < 1.0	Reducing exposure scale to less than Study Area-wide could increase the magnitude of HQs in localized areas and would still result in HQs > 1.0.
River otter	**6.0**	**1.2**			

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – low-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SUF – site use factor
TEQ – toxic equivalent
Bold identifies HQs > 1.0.

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Table 5-74. Summary of Uncertainties for Dioxin TEQs

Receptor	NOAEL HQ	LOAEL HQ	Effects Uncertainty	Exposure Uncertainty	Exposure Scale Analysis
Spotted sandpiper	0.013 – **3.2** (clam)	0.0013 – 0.32 (clam)	high to moderate: TRV is based on weekly injection study; limited toxicity data available; TEFs for birds are moderately uncertain and may over or underestimate risks of dioxins and furans for birds	high: beach evaluation assumes 100% site use of individual beach areas for breeding; shorebird areas with LOAEL HQs > 1.0 are all high habitat; FWM used to estimate tissues at five beach locations; diet based on 100% ingestion of worms may overestimate risks to shorebirds (ingestion of clams predicts no risks)	On a beach-by-beach assessment of risk, clam ingestion results in LOAEL HQs <1.0 at all beaches; worm ingestion results in LOAEL HQ > 1.0 at B16 and B17 only
	0.032 – **78** (worm)	0.0032 – **7.8** (worm)			
Hooded merganser	**4.4**	0.44		low: changing dietary portions does not result in significantly higher HQs; reducing SUF to 0.5 would not significantly reduce HQs	Reducing exposure scale to less than Study Area-wide would not significantly increase the magnitude of HQs.
Osprey	0.47	0.047		low: changing dietary portions does not result in significantly higher HQs	Reducing exposure scale to less than Study Area-wide would still result in all NOAEL and LOAEL HQs < 1.0.
Bald eagle	0.35	0.035		low: higher ingestion of peamouth would result in NOAEL HQ > 1.0, but likelihood of higher peamouth ingestion is low	
Osprey – (modeled egg)	**59**	**4.3**	moderate: TRV is based on field collected bald eagle eggs in Columbia River	moderate to high: bird egg model approach assumes literature derived BMFs, which may include chemical exposure from outside the Study Area (in areas where osprey winter) and overestimate exposure; TEFs for birds are moderately uncertain; may over or underestimate risks of dioxin-like PCBs for birds; reducing SUF to 0.5 would not significantly reduce HQs (all HQs still > 1.0)	Not evaluated.
Bald eagle (modeled egg)	**100**	**7.3**			
Mink	**18**	**3.5**	moderate: TRV is based on field collected fish with other contaminants present (i.e., PCBs)	low: changing dietary portions does not result in higher HQs; ingestion of sediment from 5-ft water depth would result in LOAEL HQ < 1.0; updated 2006 TEFs for mammals results in similar, but slightly lower HQs	Reducing exposure scale to less than Study Area-wide would not significantly increase the magnitude of HQs.
River otter	**4.7**	0.92		low to moderate: updated 2006 TEFs for mammals results in similar, but slightly lower HQs; higher ingestion of crayfish would result in LOAEL HQ > 1.0	Reducing exposure scale to less than Study Area-wide could result in LOAEL HQs > 1.0 and < 2.0 in localized areas.

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – low-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

Bold identifies HQs > 1.0.

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Table 5-75. Summary of Uncertainties for Metals Round 2 COPCs

Receptor	NOAEL HQ	LOAEL HQ	Effects Uncertainty	Exposure Uncertainty	Exposure Scale Analysis
Antimony					
Mink	0.0000049	NA	moderate: limited toxicity data available	low: changing dietary portions does not result in significantly higher HQs	Reducing exposure scale to less than Study Area-wide would still result in all NOAEL and LOAEL HQs < 1.0.
Arsenic					
Spotted sandpiper	0.24 – 0.67 (clam)	0.0081 – 0.23 (clam)	high: TRV is based on exposure to As and Cu; HQs likely overpredict bird toxicity	moderate: beach evaluation assumes 100% site use of individual beach areas for breeding; UCL used to estimate tissues at five beach locations	On a beach-by-beach assessment of risk, clam and worm ingestion results in LOAEL HQs < 1.0 at all beaches.
	0.28 – **1.6** (worm)	0.10 – 0.54 (worm)			
Cadmium					
Spotted sandpiper	0.080 – 0.38 (clam)	0.020 – 0.096 (clam)	moderate: TRV based on eggshell thinning in chickens and unknown if at these TRVs would affect other non-domestic species and reproductive success	moderate: beach evaluation assumes 100% site use of individual beach areas for breeding; UCL used to estimate worm tissue and site-specific regression used to estimate clam tissues at five beach locations	On a beach-by-beach assessment of risk, clam and worm ingestion results in LOAEL HQs < 1.0 at all beaches.
	0.064 – 0.40 (worm)	0.016 – 0.10 (worm)			
Chromium					
Spotted sandpiper	0.60 – **3.5** (clam)	0.12 – 0.71 (clam)	low: original citation could not be obtained	moderate: beach evaluation assumes 100% site use of individual beach areas for breeding; UCL used to estimate tissues at five beach locations	On a beach-by-beach assessment of risk, clam and worm ingestion results in LOAEL HQs < 1.0 at all beaches.
	0.72 – **3.4** (worm)	0.14 – 0.68 (worm)			
Copper					
Spotted sandpiper	0.095 – 0.38 (clam)	0.072 – 0.29 (clam)	low: subchronic studies available where growth was affected in birds at lower effect levels	moderate: beach evaluation assumes 100% site use of individual beach areas for breeding; UCL used to estimate tissues at five beach locations	On a beach-by-beach assessment of risk, clam and worm ingestion results in LOAEL HQs < 1.0 at all beaches.
	0.058 – 0.22 (worm)	0.044 – 0.17 (worm)			
Hooded merganser	0.04	0.027		low: changing dietary portions does not result in significantly higher HQs	Reducing exposure scale to less than Study Area-wide would still result in all NOAEL and LOAEL HQs < 1.0.
Mink	0.044	0.030	low: lowest TRVs reported in reviewed toxicological studies	low: changing dietary portions does not result in significantly higher HQs	Reducing exposure scale to less than Study Area-wide would still result in all NOAEL and LOAEL HQs < 1.0.
Lead					
Spotted sandpiper	0.10 – **3.9** (clam)	0.010 – 0.39 (clam)	low: lowest TRVs reported in reviewed toxicological studies	moderate: beach evaluation assumes 100% site use of individual beach areas for breeding; site-specific regression used to estimate tissues at five beach locations	On a beach-by-beach assessment of risk, clam and worm ingestion results in LOAEL HQs < 1.0 at all beaches.
	0.19 – **4.1** (worm)	0.019 – 0.41 (worm)			

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Table 5-75. Summary of Uncertainties for Metals Round 2 COPCs

Receptor	NOAEL HQ	LOAEL HQ	Effects Uncertainty	Exposure Uncertainty	Exposure Scale Analysis
Hooded merganser	0.44	0.044		low: higher ingestion of peamouth would result in NOAEL HQ > 1.0, but likelihood of peamouth ingestion is low	Reducing exposure scale to less than Study Area-wide could result in NOAEL HQs > 1.0 in localized areas, but LOAEL HQs would be < 1.0.
Osprey	0.052	0.0052		low: changing dietary portions does not result in significantly higher HQs	
Bald eagle	0.14	0.014		low: higher ingestion of peamouth would result in NOAEL HQ > 1.0, but likelihood of increased peamouth ingestion is low	
Mink	0.052	0.0065	low: lowest TRVs reported in reviewed toxicological studies	low: changing dietary portions does not result in significantly higher HQs	Reducing exposure scale to less than Study Area-wide would still result in all NOAEL and LOAEL HQs < 1.0.
Mercury					
Spotted sandpiper	0.78 – **2.5** (clam)	0.16 – 0.49 (clam)	moderate: other toxicological studies did not observed adverse effects at selected LOAEL TRV and LOAEL TRVs may overestimate toxicity to birds; NOAEL TRV extrapolated using a UF	moderate: beach evaluation assumes 100% site use of individual beach areas for breeding; UCL used to estimate tissues at five beach locations	On a beach-by-beach assessment of risk, clam and worm ingestion results in LOAEL HQs < 1.0 at all beaches.
	0.48 – **2.0** (worm)	0.095 – 0.40 (worm)			
Hooded merganser	**1.4**	0.28		low: changing dietary portions does not result in significantly higher HQs; reducing SUF to 0.5 would result in NOAEL HQ < 1.0	Reducing exposure scale to less than Study Area-wide could result in an increased magnitude of NOAEL HQs; however, LOAEL HQs would still be < 1.0.
Osprey	**2.1**	0.43		low: higher ingestion of northern pikeminnow would result in increased NOAEL HQ magnitude and LOAEL HQ > 1 (for osprey only), but likelihood of higher pikeminnow ingestion is low reducing SUF to 0.5 would not significantly reduce HQs	Reducing exposure scale to less than Study Area-wide could result in an increased magnitude of NOAEL HQs; however, LOAEL HQs would still be < 1.0.
Bald eagle	0.96	0.19		moderate: modeled ingestion of birds and fish would result in a NOAEL HQ > 1.0 but a LOAEL HQ < 1.0	Reducing exposure scale to less than Study Area-wide could result in NOAEL HQs > 1.0 in localized areas, but LOAEL HQs would be < 1.0.
Osprey (modeled egg)	0.60	0.41	moderate: NOAEL TRV is based on field collected eggs in various locations	moderate: bird egg model approach assumes literature derived BMFs	Not evaluated.
Bald eagle (modeled egg)	0.48	0.32			
Mink	0.77	0.16	moderate: TRV is based on field collected fish with other, unknown contaminants present	high: slightly higher ingestion of northern pikeminnow would result in NOAEL HQ > 1.0	Reducing exposure scale to less than Study Area-wide could result in NOAEL HQs > 1.0 in localized areas, but LOAEL HQs would be < 1.0.

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Table 5-75. Summary of Uncertainties for Metals Round 2 COPCs

Receptor	NOAEL HQ	LOAEL HQ	Effects Uncertainty	Exposure Uncertainty	Exposure Scale Analysis
Selenium					
Spotted sandpiper	0.23–0.70 (clam)	0.12–0.36 (clam)	low: lowest TRVs reported in reviewed toxicological studies	moderate: beach evaluation assumes 100% site use of individual beach areas for breeding; UCL used to estimate tissues at five beach locations	On a beach-by-beach assessment of risk, clam and worm ingestion results in LOAEL HQs < 1.0 at all beaches.
	0.23–1.0 (worm)	0.12–0.53 (worm)			
Mink	0.93	0.64	low: lowest TRVs reported in reviewed toxicological studies	low: higher ingestion of northern pikeminnow would result in NOAEL and LOAEL HQs > 1.0, but likelihood of increased pikeminnow ingestion is low	Reducing exposure scale to less than Study Area-wide could result in NOAEL and LOAEL HQs > 1.0
Thallium					
Spotted sandpiper	0.92 (clam)	0.018 (clam)	high: limited toxicity data available and NOAEL TRV extrapolated using a UF	moderate: beach evaluation assumes 100% site use of individual beach areas for breeding	Beach by beach assessment could not be conducted for shorebirds because no sediment data were available from beach areas.
Zinc					
Spotted sandpiper	0.18–**1.1** (clam)	0.12–0.76 (clam)	moderate: TRVs based on subchronic study; may over or underestimate chronic zinc toxicity	moderate: beach evaluation assumes 100% site use of individual beach areas for breeding; UCL used to estimate clam tissue and site-specific regression used to estimate worm tissues at five beach locations	On a beach-by-beach assessment of risk, clam and worm ingestion results in LOAEL HQs < 1.0 at all beaches.
	0.34–0.86 (worm)	0.23–0.57 (worm)			
Tributyltin ion					
Spotted sandpiper	0.00053–0.053 (clam)	0.00021–0.021 (clam)	moderate: limited toxicity data available	moderate: beach evaluation assumes 100% site use of individual beach areas for breeding; no data were available from five beach locations	On a beach-by-beach assessment of risk, clam and worm ingestion results in LOAEL HQs < 1.0 at all beaches.
	0.00035–0.045 (worm)	0.00013–0.017 (worm)			

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – low-observed-adverse-effect level
NA – not available
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

Bold identifies HQs > 1.0.

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Table 5-76. Summary of Uncertainties for Total PAHs and Benzo(a)pyrene

Receptor	NOAEL HQ	LOAEL HQ	Effects Uncertainty	Exposure Uncertainty	Exposure Scale Analysis
Total PAHs					
Spotted sandpiper	0.000021 – 0.035 (clam)	NA	very high: TRV based on petroleum hydrocarbon mixture that contained some individual PAHs in addition to other chemicals; no LOAEL TRV available; limited toxicological studies available	moderate: beach evaluation assumes 100% site use of individual beach areas for breeding; site-specific regression used to estimate tissues at five beach locations	On a beach-by-beach assessment of risk, clam and worm ingestion results in LOAEL HQs < 1.0 at all beaches.
	0.0025 – 0.680 (worm)				
Mink	0.35	0.035	high: based on benzo(a)pyrene TRVs as a surrogate for total PAHs	low: changing dietary portions does not result in significantly higher HQs	Reducing exposure scale to less than Study Area-wide would still result in all NOAEL and LOAEL HQs < 1.0.
River otter	0.10	0.010			
Benzo(a)pyrene					
Spotted sandpiper	0.00015 – 0.26 (clam)	0.000030 – 0.051 (clam)	high: TRV is based on weekly injection study and endpoints measured on effects on reproductive success are not clear	moderate: beach evaluation assumes 100% site use of individual beach areas for breeding; site-specific regression used to estimate tissues at five beach locations	On a beach-by-beach assessment of risk, clam and worm ingestion results in LOAEL HQs < 1.0 at all beaches.
	0.00016 – **4.1** (worm)	0.000032 – 0.81 (worm)			
Hooded merganser	0.13	0.025		low: changing dietary portions does not result in significantly higher HQs	Reducing exposure scale to less than Study Area-wide could result in NOAEL HQs > 1.0 in localized areas, but LOAEL HQs would be < 1.0.
Osprey	0.028	0.0056			

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – low-observed-adverse-effect level
NA – not available
Bold identifies HQs > 1.0.
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

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Table 5-77. Summary of Uncertainties for Pesticides

Receptor	NOAEL HQ	LOAEL HQ	Effects Uncertainty	Exposure Uncertainty	Exposure Scale Analysis
Aldrin					
Spotted sandpiper	0.0030 – 0.74 (clam)	0.00060 – 0.15 (clam)	moderate: limited toxicity data available and NOAEL TRV extrapolated using a UF	high: beach evaluation assumes 100% site use of individual beach areas for breeding; shorebird area with LOAEL HQ > 1.0 is a high habitat area; FWM used to estimate tissues at five beach locations; diet based on 100% ingestion of worms may overestimate risks to shorebirds (ingestion of clams predicts no risks)	On a beach-by-beach assessment of risk, clam ingestion results in LOAEL HQs <1.0 at all beaches; worm ingestion results in LOAEL HQ > 1.0 at B22.
	0.0094 – **5.4** (worm)	0.0019 – **1.1** (worm)			
Sum DDD					
Spotted sandpiper	0.0063 – **1.6** (clam)	0.0013 – 0.32 (clam)	moderate: limited toxicity data available and NOAEL TRV extrapolated using a UF	high: beach evaluation assumes 100% site use of individual beach areas for breeding; shorebird areas with LOAEL HQs > 1.0 are both high habitat; FWM used to estimate tissues at five beach locations; diet based on 100% ingestion of worms may overestimate risks to shorebirds (ingestion of clams predicts no risks)	On a beach-by-beach assessment of risk, clam ingestion results in LOAEL HQs <1 at all beaches; worm ingestion results in LOAEL HQ > 1.0 at B16 and B17.
	0.0056 – **8.2** (worm)	0.0011 – **1.6** (worm)			
Sum DDE					
Spotted sandpiper	0.013 – **2.0** (clam)	0.0027 – 0.40 (clam)	moderate: NOAEL TRV extrapolated using a UF	moderate: beach evaluation assumes 100% site use of individual beach areas for breeding; FWM used to estimate tissues at five beach locations	On a beach-by-beach assessment of risk, clam and worm ingestion results in LOAEL HQs < 1.0 at all beaches.
	0.029 – **3.5** (worm)	0.0058 – 0.70 (worm)			
Hooded merganser	**1.3**	0.26		low: changing dietary portions does not result in significantly higher HQs; reducing SUF to 0.5 would result in NOAEL HQ < 1.0	Reducing exposure scale to less than Study Area-wide could result in an increased magnitude of NOAEL HQs; however, LOAEL HQs would still be < 1.0.
Osprey (dietary dose)	0.58	0.12		low: higher ingestion of northern pikeminnow would result in NOAEL HQ > 1.0, but likelihood of pikeminnow ingestion is low	Reducing exposure scale to less than Study Area-wide would not significantly increase the magnitude of HQs.
Bald eagle (dietary dose)	0.35	0.071		moderate: modeled ingestion of birds and fish would result in a NOAEL HQ > 1.0 but a LOAEL HQ < 1.0	

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Table 5-77. Summary of Uncertainties for Pesticides

Receptor	NOAEL HQ	LOAEL HQ	Effects Uncertainty	Exposure Uncertainty	Exposure Scale Analysis
Osprey (modeled egg)	**12**	**4.4**	moderate: TRV is based on field collected bald eagle eggs in numerous locations	moderate to high: bird egg model approach assumes literature derived BMFs, which may include chemical exposure from outside the Study Area (in areas where osprey winter) and overestimate exposure; reducing SUF to 0.5 would not significantly reduce HQs (HQs still > 1.0)	Not evaluated.
Bald eagle (modeled egg)	**13**	**4.7**			
Sum DDT					
Spotted sandpiper	0.0055 – **4.8** (clam)	0.0011 – 0.95 (clam)	moderate: unknown whether eggshell thinning is a population level effect and NOAEL TRV extrapolated using a UF	moderate: beach evaluation assumes 100% site use of individual beach areas for breeding; FWM used to estimate tissues at five beach locations	On a beach-by-beach assessment of risk, clam and worm ingestion results in LOAEL HQs < 1.0 at all beaches.
	0.0027 – **1.9** (worm)	0.00054 – 0.38 (worm)			
Hooded merganser	**2.1**	0.42		low: changing dietary portions does not result in significantly higher HQs; reducing SUF to 0.5 would not significantly reduce HQs	Reducing exposure scale to less than Study Area-wide could result in LOAEL HQs > 1.0 and < 4.0 in localized areas.
Osprey	**1.7**	0.35		low: changing dietary portions does not result in significantly higher HQs; reducing SUF to 0.5 would result in NOAEL HQ < 1.0	Reducing exposure scale to less than Study Area-wide would not significantly increase the magnitude of HQs.
Bald eagle	0.69	0.14		moderate: higher ingestion of largescale sucker would result in NOAEL HQ > 1.0	
Total DDTs					
Spotted sandpiper	0.012 – **3.1** (clam)	0.0012 – 0.31 (clam)	moderate: unknown whether eggshell thinning is a population level effect	moderate: beach evaluation assumes 100% site use of individual beach areas for breeding; FWM used to estimate tissues at five beach locations	On a beach-by-beach assessment of risk, clam and worm ingestion results in LOAEL HQs < 1.0 at all beaches.
	0.012 – **9.7** (worm)	0.0012 – 0.97 (worm)			
Hooded merganser	**2.0**	0.20		low: changing dietary portions does not result in significantly higher HQs; reducing SUF to 0.5 would result in NOAEL HQ < 1.0	Reducing exposure scale to less than Study Area-wide could result in an increased magnitude of NOAEL HQs; however, LOAEL HQs would still be < 1.0.

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Table 5-77. Summary of Uncertainties for Pesticides

Receptor	NOAEL HQ	LOAEL HQ	Effects Uncertainty	Exposure Uncertainty	Exposure Scale Analysis
Mink	0.29	0.058	moderate: NOAEL TRV extrapolated using a UF	moderate: higher ingestion of sculpin would result in NOAEL HQ > 1.0	Reducing exposure scale to less than Study Area-wide would not significantly increase the magnitude of HQs.
River otter	0.24	0.048		low: higher ingestion of sculpin would result in NOAEL HQ > 1.0, but likelihood of sculpin ingestion is low	

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – low-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
Bold identifies HQs > 1.0.

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Table 5-78. Summary of Uncertainties for Phthalate COPCs

Receptor	NOAEL HQ	LOAEL HQ	Effects Uncertainty	Exposure Uncertainty	Exposure Scale Analysis
BEHP					
Spotted sandpiper	0.021 – 0.14 (clam)	0.000094 – 0.00061 (clam)	moderate: limited toxicity data available	beach evaluation assumes 100% site use of individual beach areas for breeding; UCL used to estimate tissues at five beach locations	On a beach-by-beach assessment of risk, clam and worm ingestion results in LOAEL HQs < 1.0 at all beaches.
	0.021 – 0.16 (worm)	0.000095 – 0.0071 (worm)			
Hooded merganser	**2.3**	0.010	moderate: limited toxicity data available	low: changing dietary portions does not result in significantly higher HQs; reducing SUF to 0.5 would not significantly reduce HQs	Reducing exposure scale to less than Study Area-wide could result in an increased magnitude of NOAEL HQs; however, LOAEL HQs would still be < 1.0.
Osprey	**1.6**	0.0072	moderate: limited toxicity data available	moderate: higher ingestion of northern pikeminnow and smallmouth bass would result in increased magnitude of NOAEL HQ, but likelihood of sculpin ingestion is low; reducing SUF to 0.5 would result in NOAEL HQ < 1.0	Reducing exposure scale to less than Study Area-wide would not significantly increase the magnitude of HQs.
Bald eagle	0.56	0.0025	moderate: limited toxicity data available	low: higher ingestion of northern pikeminnow would result in NOAEL HQ > 1.0, but likelihood of higher pikeminnow ingestion is low	
Dibutyl phthalate					
Spotted sandpiper	0.0065 – 1.0 (clam)	0.000029 – 0.0046 (clam)	moderate: limited toxicity data available and BEHP TRVs used as surrogate	moderate: beach evaluation assumes 100% site use of individual beach areas for breeding; UCL used to estimate tissues at five beach locations	On a beach-by-beach assessment of risk, clam and worm ingestion results in LOAEL HQs < 1.0 at all beaches.
	0.0065 – 0.39 (worm)	0.000029 – 0.0017 (worm)			

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – low-observed-adverse-effect level
NOAEL –no-observed-adverse-effect level
TRV – toxicity reference value
UCL – upper confidence limit
Bold identifies HQs > 1.0.

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Table 6-1. Amphibians and Reptiles Potentially Present Within the Study Area

Common Name	Scientific Name
Bullfrog	*Rana catesbeiana*
Common garter snake	*Thamnophis sirtalis*
Long-toed salamander	*Ambystoma macrodactylum*
Northern red-legged frog[a]	*Rana aurora*
Northwestern garter snake	*Thamnophis ordinoides*
Pacific tree frog	*Pseudacris regilla*
Painted turtle[b]	*Chrysemys picta*

[a] Federally listed as a species of concern and Oregon state listed as a vulnerable species on the ODFW sensitive species list (ODFW 2005).

[b] Oregon state listed as a critical species on the ODFW sensitive species list (ODFW 2005).

Table 6-2. Results of the 2002 Amphibian Reconnaissance Survey

General Vicinity	Reconnaissance Survey Results
Near Multnomah Channel	Observed two Northern red-legged frogs.
International Slip	Observed unidentified egg mass.
Near Kinder Morgan Linnton	No amphibians or reptiles observed.
Terminal 4/Slip 1	Observed unidentified egg mass.
Terminal 4/Slip3	Unidentifiable frog call noted.
Downstream from St. John's Bridge	No amphibians or reptiles observed.
Willamette Cove	No amphibians or reptiles observed.
Saltzman Creek, Willbridge	No amphibians or reptiles observed.
Near Gunderson	No amphibians or reptiles observed.
Swan Island Lagoon	Pacific tree frog call noted.

Table 6-3. Lines of Evidence Evaluated to Assess Risks to Amphibians

Receptor of Concern	Assessment Endpoint	Measures of Effect and Exposure (Measurement Endpoints)	Lines of Evidence
Amphibians	survival, growth, and reproduction	water exposure concentrations compared to AWQC or TRVs	Water concentrations were compared to literature-based values or AWQC to protect sensitive life stage.

AWQC – ambient water quality criteria
TRV – toxicity reference value

Table 7-1. Plant Species of the LWR

Common Name	Scientific Name	Wetland Indicator Status[a]
Alfalfa[b]	*Medicago falcata* L.	na
Bird's foot trefoil[b]	*Lotus corniculatus*	fac
Black cottonwood	*Populus balsamifera* var. *trichocarpa*	fac
Bradshaw's lomatium[c]	*Lomatium bradshawii*	facw
Canada thistle[b]	*Cirsium arvense*	fac
Cattail	*Typha latifolia*	obl
Common wetland asters[d]	*Aster* spp.	na
Columbia River willow	*Salix fluviatilis*	obl
Common groundsel[b]	*Senecio vulgaris* L.	facu
Common horsetail	*Equisetum arvense*	fac
Common rush	*Juncus effusus*	facw
Common velvet grass[b]	*Holcus lanatus* L.	fac
Douglas spiraea	*Spiraea douglasii*	facw
Himalayan blackberry[b]	*Rubus discolor*	facu
Howell's bentgrass[c]	*Agrostis howellii*	na
Hitchcock's blue-eyed grass[c]	*Sisyrinchium hitchcockii*	na
Howellia[f]	*Howellia aquatilis*	obl
Nelson's sidalcea[g]	*Sidalcea nelsonia*	fac
Oregon ash	*Fraxinus latifolia* L.	facw
Oxeye daisy[b]	*Leucanthemum vulgare*	na
Pacific willow	*Salix lucida*	facw
Peacock larkspur[c]	*Delphinium pavonaceum*	na
Piper's willow	*Salix piperii*	facw
Purple loosestrife[b]	*Lythrum salicaria*	facw
Red osier dogwood	*Cornus sericea*	facw
Reed canary grass[b]	*Phalaris arundinacea*	facw
Scots broom[b]	*Cytisus scoparius*	na
Sedge	*Carex* spp.	varies
Smartweed	*Polygonum* spp.	varies
Snowberry	*Symphoricarpos albus*	facu
St. John's wort[b]	*Hypericum perforatum*	na
Sweet clover[b]	*Melilotus alba* Mill.	na
Teasel[b]	*Dipsacus fullonum*	na
Wapato	*Sagittaria latifolia*	obl
Water moss	*Fontinalis antipyretica*	na
Wayside aster[c]	*Aster vialis*	na
White-topped aster[c]	*Aster curtus*	na

Table 7-1. Plant Species of the LWR

Common Name	Scientific Name	Wetland Indicator Status[a]
Willamette daisy[c]	*Erigeron decumbens*	na
Yellow water-flag iris[b]	*Iris pseudacorus*	obl

Sources: Adolfson et al. (2000), Integral et al. (2004)

[a] Refers to a species' fidelity to wetland environments in the Pacific Northwest (Reed 1996; Cooke and Azous 1997) and are defined as follows:

obl – obligate; high probability of occurrence in regional wetlands

facw – facultative wet; moderate to high probability of occurrence in regional wetlands

fac – facultative; moderate probability of occurrence in regional wetlands

facu – facultative upland; low to moderate probability of occurrence in regional wetlands

na – status not available

varies – status varies by species

[b] Exotic species

[c] Listed as endangered (state and federal)

[d] The aster species were garden varieties, not *Aster curtus* or *Aster vialis*

[e] Federal species of concern

[f] Listed as threatened (federal)

[g] Listed as threatened (state and federal)

Table 7-2. Lines of Evidence Evaluated to Assess Risk to Aquatic Plants

Receptor of Concern	Assessment Endpoint	Measures of Effect and Exposure (Lines of Evidence)	Lines of Evidence
Aquatic plants	survival, growth, and reproduction	bulk sediment concentrations compared to plant TRVs based on sediment exposure (if available)	Comparison of chemical concentrations in sediment to relevant plant toxicological data. TRVs from Calcasieu Estuary BERA (MacDonald ES 2002, Appendix D) were evaluated for possible relevance and use.
		surface water concentrations compared to AWQC or TRVs	Water concentrations compared to literature-based effects values or AWQC that protect sensitive life stage (e.g., germination, emergence, early life stage growth).
		TZW concentrations compared to AWQC or TRVs	Water concentrations compared to literature-based effects values or AWQC that protect sensitive life stage (e.g., germination, emergence, early life stage growth).

AWQC – ambient water quality criteria
BERA – baseline ecological risk assessment
TRV – toxicity reference value
TZW – transition zone water

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APPENDIX G: ECOLOGICAL RISK ASSESSMENT

Attachment G1
ERA Data Management

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1.0 ERA DATASET

This attachment presents the data management rules used to reduce and refine the site characterization and risk assessment (SCRA) dataset into the ecological risk assessment (ERA) dataset used in this risk analysis. Further reduction and refinement of the SCRA dataset was developed and completed by Windward Environmental LLC (Windward) for the purpose of conducting this risk analysis. The data management rules (including data reduction, data usability, and data quality) used to develop the SCRA dataset are described in detail in Section 2.0 of the Comprehensive Round 2 Site Characterization Summary and Data Gaps Analysis Report.

2.0 CALCULATED TOTALS

Calculated totals were created for analytes evaluated on the basis of summed concentrations. The calculated totals include: total polychlorinated biphenyls (PCBs) (Aroclors), total PCBs (congeners), total polycyclic aromatic hydrocarbon (PAH), total low-molecular-weight PAH (LPAH), total high-molecular-weight PAH (HPAH), sum DDD, sum DDE, sum DDT, total DDTs, total chlordane, and total endosulfan. The summation rules that were applied to all of the calculated totals are presented in Section 1.1.1, and the specific components of each calculated total are presented in Section 1.1.2.

2.1 GENERAL SUMMATION RULES

Calculated totals are the sum of all detected concentrations, and non-detected results are included based on the following rules:

- Non-detects for analytes detected at least once in the datasets defined in Section 2.0 of Appendix G were included in the summation using one-half the reporting limit (RL).
- Non-detects for analytes that were never detected within each dataset defined in Section 2.0 of Appendix G were not included in the summation (Table 2-1).
- If all of the analytes were not detected, then the highest RL was the selected value for the calculated total. The total was U-qualified to indicate the lack of detected values.

In order to ensure that the sums were always calculated based on comparable datasets, the number of components that were available were evaluated prior to summing. The number of analytes could be reduced because the sample was analyzed for a subset of the analytes (i.e., analysis of a subset of congeners rather than the complete list) or because data were removed because of quality assurance concerns.

The number of analytes required for the calculation of each value is summarized in Table 2-2. If a limited number of analytes was available, then the total was calculated and L-qualified to indicate that there were analytes unavailable for summation. If an unacceptable number of analytes was available, then the total was not calculated. Only one sediment sample was found to have an insufficient number of analytes for the calculation of total PCBs (Aroclors). That sample was analyzed for only one Aroclor. Ninety-eight sediment samples were determined to have insufficient congener data for the calculation of total PCBs (congeners). These samples were analyzed for a limited group of congeners, which represent a small subset of the total PCB concentrations for these samples. Fifty-four transition zone water samples were analyzed for one PAH compound, naphthalene, therefore, total LPAH and total PAH values were not calculated for these samples.

2.2 CALCULATED TOTALS

Total PCBs were calculated two ways: as total PCB Aroclors and as total PCB congeners. Total PCBs (Aroclors) represent the sum of Aroclors. Total PCBs (congeners) represent the sum of the 209 individual congeners. The total PCB concentration for all data in the ERA dataset is represented by total PCB Aroclors, with the exception of Round 2 tissue samples and XAD surface water samples. These Round 2 tissue samples (juvenile chinook salmon, laboratory clams, field clams, laboratory worms) and XAD surface water samples were analyzed for PCB congeners, and Aroclor results were calculated from the congener results. These calculated Aroclor results were removed from the ERA dataset because the Aroclor totals were not consistent with the total PCB values based on congener results.

Total LPAHs are the sum of concentrations for naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, and anthracene. Total HPAHs are the sum of detected concentrations for fluoranthene, pyrene, benzo(a)anthracene, chrysene, total benzofluoranthenes, benzo(a)pyrene, indeno(1,2,3,-c,d)pyrene, dibenzo(a,h) anthracene, and benzo(g,h,i)perylene. Total PAHs are the sum of LPAHs and HPAHs.

Total DDTs are the sum the six DDT isomers: 2,4′-DDD; 4,4′-DDD; 2,4′-DDE; 4,4′-DDE; 2,4′-DDT; and 4,4′-DDT. The sum DDD was the sum of 2,4′-DDD and 4,4′-DDD, the sum DDE was the sum of 2,4′-DDE and 4,4′-DDE, and the sum DDT was the sum of 2,4′-DDT and 4,4′-DDT.

Total chlordane was calculated as the sum of the following compounds: alpha-chlordane, gamma-chlordane, oxychlordane, cis-nonachlor, and trans-nonachlor.

Total endosulfan was calculated as the sum of alpha-endosulfan, beta-endosulfan, and endosulfan sulfate.

Total xylene was calculated as the sum of m,p-xylene, o-xylene, and xylene.

3.0 CALCULATION OF TOXICITY EQUIVALENTS

3.1 CALCULATION OF PCB CONGENER TEQS

PCB congener toxic equivalents (TEQs) were calculated using the World Health Organization (WHO) consensus toxic equivalency factor (TEF) values for fish, birds, and mammals (Van den Berg et al. 1998), as presented in Table 3-1. The WHO recently developed updated TEFs for mammals and PCB congeners (Van den Berg et al. 2006). A PCB TEQ for mammals based on these more recent TEQs and a comparison of the calculated TEQs on risk conclusions was evaluated in Section 5.0 of Appendix G. TEQs were calculated as the sum of each congener concentration multiplied by the corresponding TEF value. When the congener concentration was reported as non-detected, then the TEF was multiplied by half the RL.[1]

3.2 CALCULATION OF DIOXIN AND FURAN TEQS

Dioxin and furan TEQs were calculated using the WHO consensus TEF values for fish, birds, and mammals (Van den Berg et al. 1998), as presented in Table 3-2. The WHO recently developed updated TEFs for mammals and dioxins and furans (Van den Berg et al. 2006). A dioxin TEQ for mammals based on these more recent TEQs and a comparison of the calculated TEQs on risk conclusions was evaluated in Section 5.0 of Appendix G. The TEQ was calculated as the sum of each congener concentration multiplied by the corresponding TEF value. When the congener concentration was reported as non-detected, then the TEF was multiplied by half the RL.

[1] All PCB congeners were calculated at least once within both tissue and surface sediment in the ERA dataset.

4.0 SIGNIFICANT FIGURES

The laboratory reported results with different numbers of significant figures depending on the instrument, parameter, and the concentration relative to the RL. The reported (or assessed) precision of each observation was explicitly stored in the project database as a record of the number of significant figures assigned by the laboratory. The tracking of significant figures became important when calculating averages and performing other data summaries.

When a calculation involved addition, such as when totaling PCBs or PAHs, the calculation could only be as precise as the least precise number that went into the calculation. For example (assuming two significant figures):

> 210 + 19 = 229, but this would be reported as 230 because the trailing zero in the number 210 is not significant.

When a calculation involved multiplication or division, such as when carbon normalizing was used, all significant figures were carried through the calculation, and then the total result was rounded at the end of the calculation to reflect the value used in the calculation with the fewest significant figures. For example:

> 59.9 x 1.2 = 71.88, to be reported as 72 because there are two significant figures in the number 1.2.

When rounding, if the number following the last significant figure was less than 5, the digit was left unchanged. If the number following the last significant figure was equal to or greater than 5, the digit was increased by 1.

Analytical results are reported as they appeared in the SCRA database. Significant figures were not explicitly tracked in the SCRA database therefore, a result of 1.0, which has two significant figures, appeared to have only one when the significant figures were identified as non-zero integers. A minimum of 2 significant figures was used for all results. This is consistent with the standard reporting requirements of analytical laboratories. This step was necessary to ensure that summary calculations were not forced to one significant figure because of trailing zeros.

5.0 FIELD REPLICATES

Field replicates were treated as discrete samples during data analysis. A small subset of surface sediment field replicates had coordinates identical to those of the parent samples. In geographic information system (GIS) applications, such as the calculation of spatially weighted average concentrations (SWACs), the calculation requires one result for each coordinate location. In cases were the coordinates were identical, these surface sediment field replicates were excluded from GIS exports to prevent mapping errors (Table 5-1).

6.0 EXPOSURE CONCENTRATION CALCULATIONS

Exposure point concentrations (EPCs) in various media (i.e., invertebrate and fish tissue, surface sediment, and surface water) were calculated in the ERA for COPC/receptor pairs to determine risk estimates. Upper confidence limits (UCLs) were calculated to represent EPCs using ProUCL v. 3.0 (EPA 2004b). ProUCL software was developed by EPA to compute an appropriate UCL of an unknown population mean. ProUCL tests for normality, lognormality, and a gamma distribution of the dataset and computes a conservative and stable 95% UCL of the unknown population mean (EPA 2004a).

Prior to running data through ProUCL to determine UCLs, non-detected data were adjusted to one half the detection limit to represent the concentration value. For each COPC, the UCL recommended by ProUCL was used at the EPC for the risk calculations of the ERA. In the case where an insufficient number of unique concentrations values were available (where $n \leq 3$), the maximum concentration[2] was used to represent the EPC. ProUCL v. 4.0 is now available and will be used to calculate UCLs for the baseline ERA (BERA) because it provides a more sophisticated method for incorporating non-detect values.

A summary of the UCLs derived for each receptor and COPC is presented in Table 6-1[3]. Detailed descriptions of the specific statistic and distribution type calculated for each EPC, as well as summary statistics (i.e., minimum, maximum, and mean COPC concentrations) on the dataset used in the calculation are presented in Tables 6-2, 6-3, 6-4, and 6-5, for biota tissue, shorebird tissue and sediment, surface sediment, and surface water, respectively.

[2] When the maximum concentration was a non-detected value, the full detect limit was used to represent the EPC.
[3] "NJ" and "T" qualified data were included in the calculation of EPCs.

7.0 REFERENCES

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TABLES

Table 2-1. Analytes Excluded from Totals Because They Were Not Detected in ERA Dataset

Calculated Totals	Excluded Analytes
Sediment	
Total PCBs (Aroclors)	Aroclor 1232, Aroclor 1262
Total PCBs (congeners)	PCB 161
Benthic Tissue	
Total PCBs (Aroclors)	Aroclor 1016, Aroclor 1221, Aroclor 1232, Aroclor 1262, Aroclor 1268
Total PCBs (congeners)	PCB 192, PCB 161, PCB 078
Fish Tissue	
Total PCBs (Aroclors)	Aroclor 1016, Aroclor 1221, Aroclor 1254,[a] Aroclor 1262, Aroclor 1268
Total PCBs (congeners)	PCB 192, PCB 161, PCB 080, PCB 014
Surface Water	
Total PCBs (Aroclors)	Aroclor 1016, Aroclor 1221, Aroclor 1232, Aroclor 1262, Aroclor 1268
Total PCBs (congeners)	PCB 161, PCB 169
Transition Zone Water – Shallow	
Total DDTs	2,4′-DDE

[a] Aroclor 1254 was treated as a site-wide non-detect for the fish tissue dataset, regardless of the presence of NJ-qualified detections in Round 2A juvenile chinook salmon samples.

PCB – polychlorinated biphenyl

ERA – ecological risk assessment

Table 2-2. Number of Analytes Required for Each Calculated Total

Calculated Total	Maximum Number of Analytes	Limited Number of Analytes (L-qualified)	Insufficient Number of Analytes (no sum calculated)	Number of Samples with No Sum
Total PCBs (Aroclors)	7 or 9	< 7	< 2	1 sediment
Total PCBs (congeners)	209	< 150	< 100	98 sediment
Total HPAHs	10	< 10	< 5	none
Total LPAHs	7	< 7	< 3	54 TZW
Total PAHs	17	< 17	< 10	54 TZW
Sum DDD	2	< 2	NA	none
Sum DDE	2	< 2	NA	none
Sum DDT	2	< 2	NA	none
Total DDTs	6	< 6	NA	none
Total chlordane	5	< 5	NA	none
Total endosulfan	3	< 3	NA	none
Total xylene	3	< 3	NA	none

HPAH – low-molecular-weight polycyclic aromatic hydrocarbon
LPAH – high-molecular-weight polycyclic aromatic hydrocarbon
NA – not applicable
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TZW – transition zone water

Table 3-1. PCB Congener TEF Values

PCB Congener Number	TEF Values (unitless)			
	Fish	Birds	Mammals	Mammals (2006)
77	0.0001	0.05	0.0001	0.0001
81	0.0005	0.1	0.0001	0.0003
105	< 0.000005	0.0001	0.0001	0.0003
114	< 0.000005	0.0001	0.0005	0.0003
118	< 0.000005	0.00001	0.0001	0.0003
123	< 0.000005	0.00001	0.0001	0.0003
126	0.005	0.1	0.1	0.1
156	< 0.000005	0.0001	0.0005	0.0003
157	< 0.000005	0.0001	0.0005	0.0003
167	< 0.000005	0.00001	0.00001	0.0003
169	0.00005	0.001	0.01	0.03
189	< 0.000005	0.00001	0.0001	0.0003

PCB – polychlorinated biphenyl
TEF – toxic equivalency factor

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Table 3-2. Dioxin and Furan TEF Values

Dioxin and Furan Congeners	TEF Values (unitless)			
	Fish	Birds	Mammals	Mammals (2006)
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin	1	1	1	1
1,2,3,7,8-Pentachlorodibenzo-*p*-dioxin	1	1	1	1
1,2,3,4,7,8-Hexachlorodibenzo-*p*-dioxin	0.5	0.05	0.1	0.1
1,2,3,6,7,8-Hexachlorodibenzo-*p*-dioxin	0.01	0.01	0.1	0.1
1,2,3,7,8,9-Hexachlorodibenzo-*p*-dioxin	0.01	0.1	0.1	0.1
1,2,3,4,6,7,8-Heptachlorodibenzo-*p*-dioxin	0.001	< 0.001	0.01	0.01
Octachlorodibenzodioxin	NA	NA	0.0001	0.0003
2,3,7,8-Tetrachlorodibenzofuran	0.1	1	0.1	0.1
1,2,3,7,8-Pentachlorodibenzofuran	0.05	0.1	0.05	0.03
2,3,4,7,8-Pentachlorodibenzofuran	0.5	1	0.5	0.3
1,2,3,4,7,8-Hexachlorodibenzofuran	0.1	0.1	0.1	0.1
1,2,3,6,7,8-Hexachlorodibenzofuran	0.1	0.1	0.1	0.1
1,2,3,7,8,9-Hexachlorodibenzo-*p*-dioxin	0.1	0.1	0.1	0.1
2,3,4,6,7,8-Hexachlorodibenzo-*p*-dioxin	0.1	0.1	0.1	0.1
1,2,3,4,6,7,8-Heptachlorodibenzo-*p*-dioxin	0.01	0.01	0.01	0.01
1,2,3,4,7,8,9-Heptachlorodibenzo-*p*-dioxin	0.01	0.01	0.01	0.01
Octachlorodibenzofuran	0.0001	0.0001	0.0001	0.0003

NA – not available
TEF – toxic equivalency factor

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Table 5-1. Surface Sediment Field Replicates Excluded from GIS Exports

Sample Type	Sample ID
LWG field replicates	LW2-B022-2
	LW2-B025-2
	LW2-GBT006-2
	LW2-GBT027-2
Non-LWG field replicates	PSYSEA98PSY08PSY52S
	PSYSEA98PSY08PSY53S
	PSYSEA98PSY72PSY74S
	PSYSEA98PSY72PSY75S
	WLCASF97S012W4169
	WLCASF97S045W4162
	WLCDRD05PG141141Dup
	WLCDRD05PG140140Dup
	WLCDRD05PG137137Dup
	WLCDRD05PG139139Dup
	WLCDRD05PG136136Dup
	WLCDRD05PG135135Dup
	WLCDRD05PG138138Dup

GIS – geographic information system
ID – identification
LWG – Lower Willamette Group

Table 6-1. Summary of Methods Used to Calculate EPCs

Receptor	Approach	COPC	Media Type
Benthic Invertebrates			
Clams (field-collected)	critical tissue residue	cadmium, copper, zinc, TBT, total PAHs, dibutyl phthalate, total PCBs, 4,4′-DDD, total DDTs	field-collected clam tissue
Clams (laboratory-tested)	critical tissue residue	copper, TBT, total PAHs, BEHP, 4,4′-DDD, total DDTs	laboratory-tested clam tissue
Worms (laboratory-tested)	critical tissue residue	arsenic, cadmium, copper, zinc, TBT, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, pyrene, total PAHs, dibutyl phthalate, total PCBs, 4,4′-DDD, total DDTs	worm tissue
Multiplate invertebrates	critical tissue residue	copper	multiplate-collected tissue
Crayfish	critical tissue residue	copper	crayfish tissue
Benthic invertebrate community	surface water exposure	zinc (dissolved), benzo(a)anthracene, benzo(a)pyrene, 4-chloro-3-methylphenol, total PCBs, 2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDT, total DDTs	surface water (near-bottom samples only)
Fish			
Largescale sucker	critical tissue residue	BEHP, total PCBs, 4,4′-DDD, total DDTs	largescale sucker tissue
	dietary dose	cadmium, copper, TBT, benzo(a)pyrene, total PCBs, total DDTs	worm, field-collected clam tissue
			surface sediment (all samples)
White sturgeon (pre-breeding)	dietary dose	cadmium, copper, mercury, TBT, total PAHs, total PCBs, total DDTs	worm, field-collected clam tissue
			surface sediment (all samples)
Juvenile chinook salmon	critical tissue residue	zinc, 4,4′-DDD	juvenile chinook salmon tissue
	dietary dose	cadmium, copper, TBT, total PCBs	multiplate-collected invertebrate tissue
			surface sediment (all samples)
		total PCBs	chinook stomach contents tissue
Sculpin	critical tissue residue	BEHP, total PCBs, 4,4′-DDD, 4,4′-DDT, beta-HCH, total DDTs	sculpin tissue
	dietary dose	cadmium, copper, TBT, total PCBs, total DDTs	worm, field-collected clam, sculpin tissue
			surface sediment (all samples)
Peamouth	critical tissue residue	lead	peamouth tissue

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Table 6-1. Summary of Methods Used to Calculate EPCs

Receptor	Approach	COPC	Media Type
	dietary dose	cadmium, copper, TBT, total PCBs, total DDTs	multiplate invertebrate, worm, field-collected clam tissue
			surface sediment (all samples)
Smallmouth bass	critical tissue residue	BEHP, total PCBs, 4,4′-DDD, total DDTs	smallmouth bass tissue
	dietary dose	cadmium, copper, TBT, total PCBs, total DDTs	sculpin, crayfish, worm tissue
			surface sediment (all samples)
Northern pikeminnow	critical tissue residue	mercury, total PCBs, total DDTs	northern pikeminnow tissue
	dietary dose	cadmium, copper, mercury, TBT, total PCBs, total DDTs	crayfish, sculpin, worm, largescale sucker, carp, peamouth, northern pikeminnow tissue
			surface sediment (all samples)
All fish	surface water exposure	zinc (dissolved), benzo(a)anthracene, benzo(a)pyrene, 4-chloro-3-methylphenol, total PCBs, 2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDT, total DDTs	surface water
Wildlife			
Spotted sandpiper	dietary dose	arsenic, cadmium, chromium, copper, lead, mercury, selenium, thallium, zinc, butyltin, TBT, benzo(a)pyrene, total PAHs, BEHP, dibutyl-phthalate, PCB TEQ, total PCBs, dioxin TEQ, aldrin, sum DDD, sum DDE, sum DDT, total DDTs, endrin, endrin aldehyde, endrin ketone	field-collected clam and worm tissue (from within beach locations only)
			surface sediment (from within beach locations only)
Hooded merganser	dietary dose	copper, lead, mercury, zinc, benzo(a)pyrene, total PAHs, BEHP, PCB TEQ, total PCBs, dioxin TEQ, aldrin, sum DDD, sum DDE, sum DDT, total DDTs, endrin	field-collected clam, crayfish, sculpin, peamouth tissue
			surface sediment (within 20-ft water depth)
Osprey	dietary dose	lead, mercury, zinc, benzo(a)pyrene, total PAHs, BEHP, PCB TEQ, total PCBs, dioxin TEQ, sum DDE, sum DDT, endrin	largescale sucker, northern pikeminnow, carp, smallmouth bass, black crappie, brown bullhead tissue
			surface sediment (within 20-ft water depth)
Bald eagle	dietary dose	lead, mercury, zinc, benzo(a)pyrene, total PAHs, BEHP, PCB TEQ, total PCBs, dioxin TEQ, sum DDE, sum DDT	largescale sucker, carp, peamouth, northern pikeminnow tissue
			surface sediment (within 20-ft water depth)

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Table 6-1. Summary of Methods Used to Calculate EPCs

Receptor	Approach	COPC	Media Type
Mink	dietary dose	antimony, copper, lead, mercury, selenium, thallium, benzo(a)pyrene, total PAHs, PCB TEQ, total PCBs, dioxin TEQ, total DDTs	crayfish, largescale sucker, carp, sculpin, smallmouth bass, juvenile chinook salmon, peamouth, northern pikeminnow, brown bullhead, black crappie tissue
			surface sediment (within 20-ft water depth)
River otter	dietary dose	total PAHs, PCB TEQ, total PCBs, dioxin TEQ, total DDTs	crayfish, field-collected clam, carp, largescale sucker, sculpin, smallmouth bass, black crappie tissue
			surface sediment (within 20-ft water depth)

BEHP – bis(2-ethylhexyl) phthalate
EPC – exposure point concentration
COPC – chemical of potential concern
TEQ – toxic equivalent
TBT – tributyltin
PCB – polychlorinated biphenyl
PAH – polycyclic aromatic hydrocarbon

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Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Clam									
Antimony	31	52%	0.0005	0.004	0.0014	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.0022
Arsenic	31	100%	0.694	1.25	0.93	mg/kg	normal (0.05)	Student's-t UCL	0.96
Cadmium	31	100%	0.05	0.218	0.094	mg/kg	lognormal (0.05)	Student's-t UCL	0.11
Chromium	31	100%	0.4	1.05	0.65	mg/kg	normal (0.05)	Student's-t UCL	0.70
Copper	31	100%	6.85	13.5	9.6	mg/kg	normal (0.05)	Student's-t UCL	10
Lead	31	100%	0.024	0.316	0.095	mg/kg	lognormal (0.05)	95% H-UCL	0.11
Mercury	28	100%	0.005	0.016	0.0094	mg/kg	normal (0.05)	Student's-t UCL	0.01
Selenium	31	94%	0.05	0.21	0.11	mg/kg	non-parametric (0.05)	Student's-t UCL	0.13
Thallium	3	100%	0.00035	0.0007	0.00048	mg/kg	insufficient data available (n $\leq$ 3)	maximum value	0.0007
Zinc	30	100%	21.4	54	35	mg/kg	normal (0.05)	Student's-t UCL	37
Tributyltin ion	27	78%	0.9	530	28	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	220
Benzo(a)pyrene	32	91%	1.4	490	39	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	240
Naphthalene	32	12%	0.485	33	3.6	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	16
Total PAHs	32	100%	34.8	4,980	618	µg/kg	lognormal (0.05)	95% H-UCL	891
Bis(2-ethylhexyl) phthalate	31	6%	26.5	170	48	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	76
Dibutyl phthalate	31	3%	8	1,300	73	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	500
Total PCBs	34	100%	50.1	2,660	245	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	576
PCB TEQ (birds)	31	100%	3.46	71.8	13.8	pg/g	non-parametric (0.05)	95% Chebyshev UCL	25
PCB TEQ (mammals)	31	100%	0.761	12	2.89	pg/g	lognormal (0.05)	95% H-UCL	3.6
PCB TEQ (mammals 2006)	31	100%	0.534	8.56	1.94	pg/g	lognormal (0.05)	95% H-UCL	2.42
Dioxin TEQ (birds)	29	100%	0.75	32.1	4.28	pg/g	non-parametric (0.05)	95% Chebyshev UCL	9.7
Dioxin TEQ (mammals)	29	100%	0.432	6.94	1.43	pg/g	non-parametric (0.05)	95% Chebyshev UCL	2.7
Dioxin TEQ (mammals 2006)	29	100%	0.407	5.63	1.24	pg/g	non-parametric (0.05)	95% Chebyshev UCL	2.25
Aldrin	34	91%	0.126	5.07	0.41	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	1.0

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Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
2,4′-DDD	34	100%	0.394	82	7.4	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	35
2,4′-DDE	34	97%	0.0978	12.5	1.2	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	5.3
2,4′-DDT	34	97%	0.167	39	3.0	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	16
4,4′-DDD	34	100%	1.76	160	20	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	83
4,4′-DDE	34	100%	4.79	94.5	18	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	32
4,4′-DDT	34	97%	0.611	75	7.0	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	33
Sum DDD	34	100%	2.15	242	28	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	120
Sum DDE	34	100%	4.89	107	19.4	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	35
Sum DDT	34	97%	0.778	114	9.9	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	49
Total DDTs	34	100%	7.82	463	57	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	130
Endrin	34	53%	0.001575	1.2	0.077	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	0.47
Endrin aldehyde	34	3%	0.0075	0.5	0.054	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	0.28
Endrin ketone	34	35%	0.000635	0.5	0.058	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	0.31
Crayfish									
Antimony	27	93%	0.002	0.02	0.0076	mg/kg	lognormal (0.05)	95% H-UCL	0.0091
Arsenic	27	100%	0.25	0.5	0.35	mg/kg	normal (0.05)	Student's-t UCL	0.37
Cadmium	27	100%	0.007	0.036	0.018	mg/kg	normal (0.05)	Student's-t UCL	0.02
Chromium	27	100%	0.09	0.9	0.47	mg/kg	gamma distribution (0.05)	approximate gamma UCL	0.57
Copper	27	100%	10.4	17.6	14	mg/kg	normal (0.05)	Student's-t UCL	15
Lead	27	100%	0.041	1.3	0.15	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.34
Mercury	27	100%	0.02	0.041	0.028	mg/kg	normal (0.05)	Student's-t UCL	0.03
Selenium	27	0%	0.1	0.15	0.14	mg/kg	non-parametric (0.05)	Student's-t UCL	0.15
Thallium	27	100%	0.0015	0.0079	0.0033	mg/kg	lognormal (0.05)	Student's-t UCL	0.0038
Zinc	27	100%	13.7	20.3	17	mg/kg	normal (0.05)	Student's-t UCL	17
Benzo(a)pyrene	27	0%	8.5	18.5	16	µg/kg	non-parametric (0.05)	Student's-t UCL	17
Naphthalene	27	0%	8.5	41	27	µg/kg	non-parametric (0.05)	Student's-t UCL	30

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Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Total PAHs	27	11%	17	731	85	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	410
Bis(2-ethylhexyl) phthalate	27	0%	49.5	135	64	µg/kg	non-parametric (0.05)	Student's-t UCL	72
Dibutyl phthalate	27	0%	165	325	180	µg/kg	non-parametric (0.05)	Student's-t UCL	190
Total PCBs	27	44%	0.85	335	37	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	170
PCB TEQ (birds)	10	100%	1.82	54.6	11.8	pg/g	gamma distribution (0.05)	approximate gamma UCL	23
PCB TEQ (mammals)	10	100%	0.491	4.55	1.79	pg/g	gamma distribution (0.05)	approximate gamma UCL	2.95
PCB TEQ (mammals 2006)	10	100%	0.252	3.13	1.14	pg/g	gamma distribution (0.05)	approximate gamma UCL	1.93
Dioxin TEQ (birds)	10	100%	1.33	105	12.9	pg/g	non-parametric (0.05)	99% Chebyshev UCL	115
Dioxin TEQ (mammals)	10	100%	0.462	22.7	3.22	pg/g	non-parametric (0.05)	99% Chebyshev UCL	25
Dioxin TEQ (mammals 2006)	10	100%	0.423	18.2	2.66	pg/g	non-parametric (0.05)	99% Chebyshev UCL	19.9
Aldrin	27	0%	0.5	1	0.52	µg/kg	non-parametric (0.05)	Student's-t UCL	0.55
2,4′-DDD	27	4%	0.5	4.3	0.66	µg/kg	non-parametric (0.05)	Student's-t UCL	0.90
2,4′-DDE	27	11%	0.5	3.3	0.67	µg/kg	non-parametric (0.05)	Student's-t UCL	0.86
2,4′-DDT	27	70%	0.5	9.5	2.6	µg/kg	gamma distribution (0.05)	approximate gamma UCL	3.3
4,4′-DDD	27	19%	0.5	17	1.7	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	4.7
4,4′-DDE	27	100%	1.6	51	6.3	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	14
4,4′-DDT	27	30%	0.5	14	2.2	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	8.7
Sum DDD	27	19%	0.50	21.3	1.9	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	10
Sum DDE	27	100%	2.1	51.5	6.9	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	15
Sum DDT	27	74%	0.50	17.5	4.6	µg/kg	lognormal (0.05)	95% H-UCL	6.8
Total DDTs	27	100%	4.6	85.4	14	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	27
Endrin	27	19%	0.5	2.8	0.74	µg/kg	non-parametric (0.05)	Student's-t UCL	0.91
Endrin aldehyde	27	0%	0.5	1	0.52	µg/kg	non-parametric (0.05)	Student's-t UCL	0.55
Endrin ketone	27	0%	0.5	1	0.52	µg/kg	non-parametric (0.05)	Student's-t UCL	0.55
Laboratory Clam									
Antimony	35	49%	0.00025	0.00175	0.00066	mg/kg	gamma distribution (0.05)	approximate gamma UCL	0.00076

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Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Arsenic	35	100%	0.303	0.548	0.43	mg/kg	lognormal (0.05)	Student's-t UCL	0.44
Cadmium	35	100%	0.0383	0.0666	0.051	mg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	0.053
Chromium	35	100%	0.14	0.49	0.22	mg/kg	non-parametric (0.05)	Student's-t UCL	0.24
Copper	35	100%	2.64	5.94	3.8	mg/kg	gamma distribution (0.05)	approximate gamma UCL	4.0
Lead	35	100%	0.033	0.221	0.058	mg/kg	non-parametric (0.05)	Student's-t UCL	0.068
Mercury	35	100%	0.00705	0.0162	0.010	mg/kg	normal (0.05)	Student's-t UCL	0.011
Selenium	35	100%	0.072	0.247	0.14	mg/kg	non-parametric (0.05)	Student's-t UCL	0.16
Zinc	35	100%	10.8	16.8	14	mg/kg	normal (0.05)	Student's-t UCL	14
Tributyltin ion	35	26%	0.445	680	22	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	210
Benzo(a)pyrene	35	94%	0.06	33	2.3	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	12
Naphthalene	35	3%	0.165	5.1	0.54	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	1.1
Total PAHs	35	100%	19.2	1,320	139	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	600
Bis(2-ethylhexyl) phthalate	35	77%	26.5	8,600	320	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	1400
Dibutyl phthalate	35	0%	8	105	51	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	120
Total PCBs	35	100%	19.1	189	43.1	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	71.0
PCB TEQ (birds)	35	100%	1.3	27.7	4.08	pg/g	non-parametric (0.05)	95% Chebyshev UCL	8.6
PCB TEQ (mammals)	35	100%	0.39	2.28	0.781	pg/g	non-parametric (0.05)	Student's-t UCL	0.91
PCB TEQ (mammals 2006)	35	100%	0.201	1.59	0.507	pg/g	non-parametric (0.05)	95% Chebyshev UCL	0.743
Dioxin TEQ (birds)	35	100%	0.23	87.7	3.35	pg/g	non-parametric (0.05)	99% Chebyshev UCL	28
Dioxin TEQ (mammals)	35	100%	0.155	42.5	1.60	pg/g	non-parametric (0.05)	99% Chebyshev UCL	14
Dioxin TEQ (mammals 2006)	35	100%	0.147	40.6	1.50	pg/g	non-parametric (0.05)	99% Chebyshev UCL	13
Aldrin	35	80%	0.00625	2.14	0.084	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.35
2,4′-DDD	35	100%	0.094	239	7.7	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	76
2,4′-DDE	35	86%	0.0125	2.93	0.15	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	1.0
2,4′-DDT	35	71%	0.01055	0.729	0.056	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.14
4,4′-DDD	35	100%	0.375	702	24	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	220

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Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
4,4′-DDE	35	100%	0.513	8.55	1.6	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	3.1
4,4′-DDT	35	91%	0.0449	87.1	2.7	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	27
Sum DDD	35	100%	0.511	941	31	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	300
Sum DDE	35	100%	0.526	11.2	1.75	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	3.58
Sum DDT	35	91%	0.0449	87.1	2.73	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	27.5
Total DDTs	35	100%	1.13	1,040	36.0	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	331
Endrin	35	14%	0.001	0.0616	0.010	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.018
Endrin aldehyde	35	0%	0.0005	0.02015	0.0097	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.012
Endrin ketone	35	9%	0.000525	0.04195	0.019	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.026
Laboratory Worm									
Antimony	35	37%	0.0005	0.014	0.0024	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.0044
Arsenic	35	100%	0.285	3.04	1.2	mg/kg	normal (0.05)	Student's-t UCL	1.4
Cadmium	35	100%	0.0364	0.254	0.069	mg/kg	non-parametric (0.05)	Student's-t UCL	0.083
Chromium	35	100%	0.14	0.89	0.51	mg/kg	normal (0.05)	Student's-t UCL	0.56
Copper	35	100%	1.83	20.2	2.9	mg/kg	non-parametric (0.05)	Student's-t UCL	3.8
Lead	35	100%	0.167	0.847	0.35	mg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	0.40
Mercury	34	100%	0.003	0.0105	0.0054	mg/kg	lognormal (0.05)	Student's-t UCL	0.006
Selenium	35	100%	0.08	0.37	0.21	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.29
Zinc	35	100%	18.2	31.5	26	mg/kg	normal (0.05)	Student's-t UCL	27
Tributyltin ion	35	43%	0.22	1,700	53	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	540
Benzo(a)anthracene	35	100%	5.9	2600	200	ug/kg	non-parametric (0.05)	99% Chebyshev UCL	1,000
Benzo(a)pyrene	35	100%	2.2	1,500	100	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	610
Benzo(b)fluoranthene	35	100%	7.7	1600	150	ug/kg	non-parametric (0.05)	99% Chebyshev UCL	760
Benzo(k)fluoranthene	35	100%	3.9	1500	110	ug/kg	non-parametric (0.05)	99% Chebyshev UCL	610
Chrysene	35	100%	14	3900	350	ug/kg	non-parametric (0.05)	99% Chebyshev UCL	1,700
Pyrene	35	100%	16	11000	870	ug/kg	non-parametric (0.05)	99% Chebyshev UCL	4,600

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Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Naphthalene	35	17%	0.23	31	2.4	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	12
Total PAHs	35	100%	83.6	37,300	3,120	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	15,960
Bis(2-ethylhexyl) phthalate	35	57%	26.5	220	95	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	140
Dibutyl phthalate	35	11%	8	470	72	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	230
Total PCBs	35	100%	44.8	4,310	612	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	2,390
PCB TEQ (birds)	35	100%	3.81	732	59.5	pg/g	non-parametric (0.05)	99% Chebyshev UCL	310
PCB TEQ (mammals)	35	100%	0.935	49.1	6.71	pg/g	non-parametric (0.05)	99% Chebyshev UCL	23
PCB TEQ (mammals 2006)	35	100%	0.676	32.4	4.52	pg/g	non-parametric (0.05)	99% Chebyshev UCL	15.4
Dioxin TEQ (birds)	35	100%	1.75	932	37.1	pg/g	non-parametric (0.05)	99% Chebyshev UCL	300
Dioxin TEQ (mammals)	35	100%	1.1	472	18.0	pg/g	non-parametric (0.05)	99% Chebyshev UCL	150
Dioxin TEQ (mammals 2006)	35	100%	1.02	448	16.6	pg/g	non-parametric (0.05)	99% Chebyshev UCL	143
Aldrin	35	97%	0.043	37	1.5	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	12
2,4′-DDD	35	100%	0.611	204	15	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	92
2,4′-DDE	35	100%	0.127	52.7	2.9	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	18
2,4′-DDT	35	66%	0.00955	3.13	0.25	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	1.3
4,4′-DDD	35	100%	4.08	1,060	71	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	420
4,4′-DDE	35	100%	9.52	171	29	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	54
4,4′-DDT	35	94%	0.0456	37.9	2.5	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	16
Sum DDD	35	100%	4.78	1,260	86.5	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	513
Sum DDE	35	100%	9.66	188	32	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	59
Sum DDT	35	97%	0.0545	40.0	2.72	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	17.3
Total DDTs	35	100%	14.5	1,490	121	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	613
Endrin	35	26%	0.001855	0.157	0.016	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.035
Endrin aldehyde	35	0%	0.000346	0.0192	0.0091	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.012
Endrin ketone	35	11%	0.00073	0.04	0.019	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.026

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Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Multiplate Invertebrates									
Antimony	2	50%	0.00055	0.0022	0.0014	mg/kg	insufficient data available (n ≤ 3)	maximum value	0.0022
Arsenic	2	100%	0.349	0.45	0.40	mg/kg	insufficient data available (n ≤ 3)	maximum value	0.45
Cadmium	2	100%	0.0321	0.0366	0.0344	mg/kg	insufficient data available (n ≤ 3)	maximum value	0.0366
Chromium	2	100%	0.64	1.73	1.2	mg/kg	insufficient data available (n ≤ 3)	maximum value	1.73
Copper	2	100%	3.01	6	4.5	mg/kg	insufficient data available (n ≤ 3)	maximum value	6.0
Lead	2	100%	0.245	1.06	0.653	mg/kg	insufficient data available (n ≤ 3)	maximum value	1.06
Selenium	2	100%	0.04	0.06	0.050	mg/kg	insufficient data available (n ≤ 3)	maximum value	0.06
Zinc	2	100%	12.6	24.8	18.7	mg/kg	insufficient data available (n ≤ 3)	maximum value	24.8
Total PCBs	7	100%	33.2	498	112	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	393
PCB TEQ (birds)	7	100%	1.39	4.98	3.03	pg/g	normal (0.05)	Student's-t UCL	3.91
PCB TEQ (mammals)	7	100%	0.439	1.88	0.808	pg/g	gamma distribution (0.05)	approximate gamma UCL	1.24
PCB TEQ (mammals 2006)	7	100%	0.289	0.767	0.463	pg/g	normal (0.05)	Student's-t UCL	0.589
Dioxin TEQ (birds)	7	100%	0.671	15.5	3.02	pg/g	non-parametric (0.05)	99% Chebyshev UCL	24
Dioxin TEQ (mammals)	7	100%	0.398	4.02	1.20	pg/g	gamma distribution (0.05)	approximate gamma UCL	2.5
Dioxin TEQ (mammals 2006)	7	100%	0.388	3.35	1.08	pg/g	gamma distribution (0.05)	approximate gamma UCL	2.13
Aldrin	7	86%	0.0064	0.0872	0.041	µg/kg	normal (0.05)	Student's-t UCL	0.065
2,4′-DDD	7	100%	0.136	14.1	2.45	µg/kg	lognormal (0.05)	99% Chebyshev (MVUE) UCL	10.7
2,4′-DDE	7	100%	0.0382	1.28	0.26	µg/kg	lognormal (0.05)	95% Chebyshev (MVUE) UCL	0.59
2,4′-DDT	7	71%	0.0086	7.54	1.2	µg/kg	lognormal (0.05)	99% Chebyshev (MVUE) UCL	5.3
4,4′-DDD	7	100%	0.646	30.3	5.44	µg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	20.7
4,4′-DDE	7	100%	1.21	29.4	7.19	µg/kg	gamma distribution (0.05)	approximate gamma UCL	18.9
4,4′-DDT	7	100%	0.0502	12.2	1.90	µg/kg	lognormal (0.05)	99% Chebyshev (MVUE) UCL	7.26
Sum DDD	7	100%	0.782	44.4	7.9	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	69

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Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Sum DDE	7	100%	1.28	30.7	7.45	µg/kg	gamma distribution (0.05)	approximate gamma UCL	19.7
Sum DDT	7	100%	0.0588	19.7	3.06	µg/kg	lognormal (0.05)	99% Chebyshev (MVUE) UCL	12.1
Total DDTs	7	100%	2.67	94.8	18.4	µg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	62
Endrin	7	14%	0.002	0.073	0.018	µg/kg	gamma distribution (0.05)	approximate gamma UCL	0.049
Endrin aldehyde	7	0%	0.002755	0.0545	0.0140	µg/kg	gamma distribution (0.05)	approximate gamma UCL	0.0339
Endrin ketone	7	0%	0.00162	0.114	0.027	µg/kg	gamma distribution (0.05)	approximate gamma UCL	0.089
Black Crappie									
Antimony	4	0%	0.0005	0.0005	0.0005	mg/kg	no distinct values in dataset[c]	maximum value	0.005
Arsenic	4	100%	0.185	0.42	0.28	mg/kg	normal (0.05)	Student's-t UCL	0.40
Cadmium	4	100%	0.003	0.006	0.0039	mg/kg	gamma distribution (0.05)	approximate gamma UCL	0.0067
Chromium	4	0%	0.04	0.045	0.041	mg/kg	non-parametric (0.05)	Student's-t UCL	0.044
Copper	4	100%	0.688	0.946	0.820	mg/kg	normal (0.05)	Student's-t UCL	0.977
Lead	4	25%	0.002	0.0185	0.0066	mg/kg	gamma distribution (0.05)	approximate gamma UCL	0.038
Mercury	4	100%	0.033	0.044	0.039	mg/kg	normal (0.05)	Student's-t UCL	0.046
Selenium	4	0%	0.15	0.15	0.15	mg/kg	no distinct values in dataset[c]	maximum value	0.30
Thallium	4	100%	0.006	0.0167	0.011	mg/kg	normal (0.05)	Student's-t UCL	0.016
Zinc	4	100%	14.2	16.8	15.4	mg/kg	normal (0.05)	Student's-t UCL	16.7
Total PCBs	4	100%	86.9	260	137	µg/kg	gamma distribution (0.05)	approximate gamma UCL	333
PCB TEQ (birds)	4	100%	10.6	41.2	18.8	pg/g	non-parametric (0.05)	95% Chebyshev UCL	51
PCB TEQ (mammals)	4	100%	2.54	5.26	3.38	pg/g	gamma distribution (0.05)	approximate gamma UCL	5.85
PCB TEQ (mammals 2006)	4	100%	1.67	3.23	2.17	pg/g	normal (0.05)	Student's-t UCL	3.03
Dioxin TEQ (birds)	4	100%	2.39	2.7	2.54	pg/g	normal (0.05)	Student's-t UCL	2.7
Dioxin TEQ (mammals)	4	100%	1.16	1.33	1.24	pg/g	normal (0.05)	Student's-t UCL	1.33
Dioxin TEQ (mammals 2006)	4	100%	1.1	1.26	1.18	pg/g	normal (0.05)	Student's-t UCL	1.26
Aldrin	4	0%	0.5	0.65	0.54	µg/kg	non-parametric (0.05)	Student's-t UCL	0.63
2,4′-DDD	4	0%	1	1.85	1.5	µg/kg	normal (0.05)	Student's-t UCL	1.9

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Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
2,4′-DDE	4	0%	0.5	0.55	0.53	µg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	0.57
2,4′-DDT	4	75%	3.45	6.6	4.9	µg/kg	normal (0.05)	Student's-t UCL	6.5
4,4′-DDD	4	100%	7.9	18.5	12	µg/kg	normal (0.05)	Student's-t UCL	17
4,4′-DDE	4	100%	37	80.5	56	µg/kg	normal (0.05)	Student's-t UCL	81
4,4′-DDT	4	75%	3.15	15	9.2	µg/kg	normal (0.05)	Student's-t UCL	15
Sum DDD	4	100%	9.45	20	14	µg/kg	normal (0.05)	Student's-t UCL	19
Sum DDE	4	100%	37.6	81	56	µg/kg	normal (0.05)	Student's-t UCL	82
Sum DDT	4	75%	3.5	21.6	13	µg/kg	normal (0.05)	Student's-t UCL	22
Total DDTs	4	100%	61.3	108	83.9	µg/kg	normal (0.05)	Student's-t UCL	112
Endrin	4	0%	0.5	0.65	0.56	µg/kg	normal (0.05)	Student's-t UCL	0.65
Endrin aldehyde	4	0%	0.5	0.7	0.56	µg/kg	normal (0.05)	Student's-t UCL	0.67
Endrin ketone	4	0%	0.5	0.5	0.50	µg/kg	no distinct values in dataset[c]	maximum value	1.0
Brown Bullhead									
Antimony	6	0%	0.0005	0.0005	0.0005	mg/kg	no distinct values in dataset[c]	maximum value	0.005
Arsenic	6	100%	0.04	0.08	0.056	mg/kg	normal (0.05)	Student's-t UCL	0.068
Cadmium	6	100%	0.008	0.014	0.012	mg/kg	normal (0.05)	Student's-t UCL	0.014
Chromium	6	100%	0.39	1.32	0.73	mg/kg	normal (0.05)	Student's-t UCL	1.0
Copper	6	100%	0.586	0.798	0.69	mg/kg	normal (0.05)	Student's-t UCL	0.76
Lead	6	83%	0.007	0.0435	0.025	mg/kg	normal (0.05)	Student's-t UCL	0.035
Mercury	6	100%	0.025	0.054	0.037	mg/kg	normal (0.05)	Student's-t UCL	0.046
Selenium	6	33%	0.1	0.3	0.18	mg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	0.3
Thallium	6	67%	0.0009	0.0039	0.0023	mg/kg	normal (0.05)	Student's-t UCL	0.0033
Zinc	6	100%	12.7	15.6	14.1	mg/kg	normal (0.05)	Student's-t UCL	15
Benzo(a)pyrene	6	0%	16.5	16.5	17	µg/kg	no distinct values in dataset[c]	maximum value	33
Naphthalene	6	0%	16.5	16.5	17	µg/kg	no distinct values in dataset[c]	maximum value	33
Total PAHs	6	17%	17	348	74	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	620

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Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Bis(2-ethylhexyl) phthalate	6	17%	49	2,700	490	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	4900
Dibutyl phthalate	6	0%	165	165	170	µg/kg	no distinct values in dataset[c]	maximum value	330
Total PCBs	6	100%	68.9	1,760	428	µg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	1460
PCB TEQ (birds)	6	100%	3.69	9.73	7.25	pg/g	normal (0.05)	Student's-t UCL	9.4
PCB TEQ (mammals)	6	100%	2.21	16.5	6.82	pg/g	normal (0.05)	Student's-t UCL	11.1
PCB TEQ (mammals 2006)	6	100%	1.6	6.04	3.47	pg/g	normal (0.05)	Student's-t UCL	4.84
Dioxin TEQ (birds)	6	100%	2.14	3.35	2.53	pg/g	normal (0.05)	Student's-t UCL	2.89
Dioxin TEQ (mammals)	6	100%	1.44	2.43	1.76	pg/g	normal (0.05)	Student's-t UCL	2.06
Dioxin TEQ (mammals 2006)	6	100%	1.29	2.12	1.58	pg/g	normal (0.05)	Student's-t UCL	1.83
Aldrin	6	0%	0.5	6.5	1.8	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	12
2,4′-DDD	6	17%	0.8	12	3.5	µg/kg	gamma distribution (0.05)	approximate gamma UCL	10
2,4′-DDE	6	0%	0.5	3.75	1.4	µg/kg	gamma distribution (0.05)	approximate gamma UCL	3.2
2,4′-DDT	6	83%	3.3	14	8.2	µg/kg	normal (0.05)	Student's-t UCL	12
4,4′-DDD	6	100%	6.9	13	9.4	µg/kg	normal (0.05)	Student's-t UCL	11
4,4′-DDE	6	100%	29.5	70	47	µg/kg	normal (0.05)	Student's-t UCL	60
4,4′-DDT	6	83%	3.15	46	20	µg/kg	normal (0.05)	Student's-t UCL	33
Sum DDD	6	100%	7.7	25	13	µg/kg	normal (0.05)	Student's-t UCL	18
Sum DDE	6	100%	32.5	70.8	49	µg/kg	normal (0.05)	Student's-t UCL	61
Sum DDT	6	83%	3.5	58	27	µg/kg	normal (0.05)	Student's-t UCL	44
Total DDTs	6	100%	48.8	143	89.5	µg/kg	normal (0.05)	Student's-t UCL	119
Endrin	6	0%	0.5	10	2.4	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	18
Endrin aldehyde	6	0%	0.5	4.25	1.7	µg/kg	normal (0.05)	Student's-t UCL	2.8
Endrin ketone	6	0%	0.5	3.75	1.3	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	3.7
Carp									
Antimony	6	0%	0.0005	0.002	0.0012	mg/kg	normal (0.05)	Student's-t UCL	0.0016

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Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Arsenic	6	100%	0.125	0.22	0.17	mg/kg	normal (0.05)	Student's-t UCL	0.20
Cadmium	6	100%	0.048	0.108	0.069	mg/kg	normal (0.05)	Student's-t UCL	0.086
Chromium	6	100%	0.305	2.02	1.1	mg/kg	normal (0.05)	Student's-t UCL	1.6
Copper	6	100%	1.04	1.42	1.16	mg/kg	normal (0.05)	Student's-t UCL	1.29
Lead	6	100%	0.121	0.202	0.15	mg/kg	normal (0.05)	Student's-t UCL	0.18
Mercury	6	100%	0.029	0.047	0.040	mg/kg	normal (0.05)	Student's-t UCL	0.045
Selenium	6	100%	0.3	0.4	0.32	mg/kg	non-parametric (0.05)	Student's-t UCL	0.35
Thallium	6	100%	0.00125	0.0049	0.0029	mg/kg	normal (0.05)	Student's-t UCL	0.0041
Zinc	6	100%	87.1	112	99	mg/kg	normal (0.05)	Student's-t UCL	110
Benzo(a)pyrene	6	0%	15.5	16.5	16	µg/kg	normal (0.05)	Student's-t UCL	16
Naphthalene	6	33%	16	56	28	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	60
Total PAHs	6	33%	16	424	140	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	930
Bis(2-ethylhexyl) phthalate	6	0%	47	49.5	48	µg/kg	normal (0.05)	Student's-t UCL	49
Dibutyl phthalate	6	0%	310	330	320	µg/kg	normal (0.05)	Student's-t UCL	330
Total PCBs	6	100%	240	7,460	1,860	µg/kg	gamma distribution (0.05)	approximate gamma UCL	6470
PCB TEQ (birds)	6	100%	13.6	34.8	21.1	pg/g	normal (0.05)	Student's-t UCL	28
PCB TEQ (mammals)	6	100%	5.39	38.8	13.2	pg/g	lognormal (0.05)	95% H-UCL	35.9
PCB TEQ (mammals 2006)	6	100%	3.36	16.2	6.86	pg/g	gamma distribution (0.05)	approximate gamma UCL	12.1
Dioxin TEQ (birds)	6	100%	3.86	18.9	7.60	pg/g	gamma distribution (0.05)	approximate gamma UCL	14
Dioxin TEQ (mammals)	6	100%	2.17	11.1	4.79	pg/g	normal (0.05)	Student's-t UCL	7.53
Dioxin TEQ (mammals 2006)	6	100%	1.98	8.53	4.15	pg/g	normal (0.05)	Student's-t UCL	6.15
Aldrin	6	0%	0.5	6.5	2.7	µg/kg	normal (0.05)	Student's-t UCL	4.8
2,4′-DDD	6	50%	4.3	81.5	22	µg/kg	gamma distribution (0.05)	approximate gamma UCL	74
2,4′-DDE	6	0%	0.5	3.75	2.2	µg/kg	normal (0.05)	Student's-t UCL	3.4
2,4′-DDT	6	17%	3.45	23	6.7	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	21
4,4′-DDD	6	100%	22.6	89.5	46	µg/kg	normal (0.05)	Student's-t UCL	67

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Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
4,4′-DDE	6	100%	81	260	140	µg/kg	normal (0.05)	Student's-t UCL	190
4,4′-DDT	6	17%	3.15	24	6.6	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	22
Sum DDD	6	100%	26.9	171	69	µg/kg	gamma distribution (0.05)	approximate gamma UCL	130
Sum DDE	6	100%	81.5	264	138	µg/kg	normal (0.05)	Student's-t UCL	192
Sum DDT	6	17%	3.5	47	11	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	83
Total DDTs	6	100%	154	441	220	µg/kg	non-parametric (0.05)	Student's-t UCL	310
Endrin	6	0%	0.5	10	3.3	µg/kg	normal (0.05)	Student's-t UCL	6.3
Endrin aldehyde	6	0%	0.9	4.25	2.4	µg/kg	normal (0.05)	Student's-t UCL	3.6
Endrin ketone	6	0%	0.5	3.75	2.1	µg/kg	normal (0.05)	Student's-t UCL	3.3
Juvenile Chinook Salmon (whole body tissue)									
Antimony	15	0%	0.0005	0.002145	0.00087	mg/kg	non-parametric (0.05)	Mod-t UCL (adjusted for skewness)	0.0011
Arsenic	15	100%	0.04646	0.25	0.11	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.19
Cadmium	15	100%	0.009653	0.027	0.014	mg/kg	non-parametric (0.05)	Student's-t UCL	0.017
Chromium	15	20%	0.027	0.19	0.050	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.10
Copper	15	100%	0.75548	2.15	1.1	mg/kg	gamma distribution (0.05)	approximate gamma UCL	1.3
Lead	15	20%	0.003	0.03705	0.014	mg/kg	gamma distribution (0.05)	approximate gamma UCL	0.019
Mercury	15	100%	0.01014	0.02	0.014	mg/kg	non-parametric (0.05)	Student's-t UCL	0.016
Selenium	15	60%	0.072	0.273	0.16	mg/kg	normal (0.05)	Student's-t UCL	0.18
Thallium	6	100%	0.005	0.0105	0.0068	mg/kg	normal (0.05)	Student's-t UCL	0.0086
Zinc	15	100%	24.038	33.3	29.5	mg/kg	normal (0.05)	Student's-t UCL	30.7
Tributyltin ion	8	100%	1.3	4.1	2.5	µg/kg	normal (0.05)	Student's-t UCL	3.2
Benzo(a)pyrene	15	0%	0.065	16.5	6.5	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	27
Naphthalene	15	53%	1.45	33	10	µg/kg	gamma distribution (0.05)	approximate gamma UCL	15
Total PAHs	15	67%	11.9	297	35	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	120
Bis(2-ethylhexyl) phthalate	11	0%	47.5	430	160	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	350

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Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Dibutyl phthalate	11	18%	15.5	260	150	µg/kg	normal (0.05)	Student's-t UCL	200
Total PCBs	15	100%	31.9	277	112	µg/kg	gamma distribution (0.05)	approximate gamma UCL	152
PCB TEQ (birds)	9	100%	4.31	10.1	7.08	pg/g	normal (0.05)	Student's-t UCL	8.3
PCB TEQ (mammals)	9	100%	1.88	4.59	2.89	pg/g	normal (0.05)	Student's-t UCL	3.44
PCB TEQ (mammals 2006)	9	100%	1.4	3.9	2.24	pg/g	normal (0.05)	Student's-t UCL	2.73
Dioxin TEQ (birds)	9	100%	1.64	8.77	4.05	pg/g	non-parametric (0.05)	95% Chebyshev UCL	8.6
Dioxin TEQ (mammals)	9	100%	1.44	4.63	2.30	pg/g	gamma distribution (0.05)	approximate gamma UCL	3.1
Dioxin TEQ (mammals 2006)	9	100%	1.37	4.38	2.14	pg/g	gamma distribution (0.05)	approximate gamma UCL	2.85
Aldrin	15	0%	0.08	0.6	0.31	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.54
2,4′-DDD	15	47%	0.5	13	3.6	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	15
2,4′-DDE	15	0%	0.08	3.1	0.68	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	2.5
2,4′-DDT	15	67%	1.35	10	4.8	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	8.7
4,4′-DDD	15	100%	1	130	27	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	150
4,4′-DDE	15	100%	7	93	30	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	66
4,4′-DDT	15	87%	4.2	38	15	µg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	21
Sum DDD	15	100%	2.15	143	31	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	170
Sum DDE	15	100%	7.75	94.1	31	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	67
Sum DDT	15	93%	6.1	48	19	µg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	26
Total DDTs	15	100%	17.5	285	81	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	190
Endrin	15	0%	0.038	0.55	0.36	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	0.95
Endrin aldehyde	15	0%	0.095	1.75	0.60	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	1.0
Endrin ketone	15	0%	0.205	1.5	0.50	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.87
Juvenile Chinook Salmon (stomach contents tissue)									
Total PAHs	5	100%	118	2,480	711	µg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	3,000
Total PCBs	5	100%	53.9	162	84.0	µg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	148
Total DDTs	5	100%	8.88	327	107	µg/kg	normal (0.05)	Student's-t UCL	242

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Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Largescale Sucker									
Antimony	6	100%	0.001	0.003	0.0023	mg/kg	Data are normal (0.05)	Student's-t UCL	0.0029
Arsenic	6	100%	0.18	0.27	0.23	mg/kg	normal (0.05)	Student's-t UCL	0.26
Cadmium	6	100%	0.015	0.0325	0.021	mg/kg	normal (0.05)	Student's-t UCL	0.026
Chromium	6	100%	0.38	2.77	1.1	mg/kg	normal (0.05)	Student's-t UCL	1.9
Copper	6	100%	0.735	1.1	0.90	mg/kg	normal (0.05)	Student's-t UCL	1.0
Lead	6	100%	0.064	0.191	0.12	mg/kg	normal (0.05)	Student's-t UCL	0.16
Mercury	6	100%	0.045	0.085	0.068	mg/kg	normal (0.05)	Student's-t UCL	0.081
Selenium	6	0%	0.15	0.15	0.15	mg/kg	no distinct values in dataset[c]	maximum value	0.30
Thallium	6	100%	0.0015	0.0038	0.0025	mg/kg	normal (0.05)	Student's-t UCL	0.0032
Zinc	6	100%	17.1	19.7	17.9	mg/kg	normal (0.05)	Student's-t UCL	18.7
Benzo(a)pyrene	6	0%	13	16.5	14	µg/kg	normal (0.05)	Student's-t UCL	16
Naphthalene	6	17%	13	54	21	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	50
Total PAHs	6	33%	13	349	120	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	770
Bis(2-ethylhexyl) phthalate	6	33%	39	3,000	660	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	5500
Dibutyl phthalate	6	0%	130	260	160	µg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	210
Total PCBs	6	100%	96.9	2,060	903	µg/kg	normal (0.05)	Student's-t UCL	1540
Aldrin	6	0%	0.5	6.5	2.5	µg/kg	lognormal (0.05)	95% H-UCL	10
2,4′-DDD	6	33%	3.9	40	12	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	38
2,4′-DDE	6	0%	0.5	3.75	2.3	µg/kg	normal (0.05)	Student's-t UCL	3.2
2,4′-DDT	6	50%	3.45	50	14	µg/kg	gamma distribution (0.05)	approximate gamma UCL	42
4,4′-DDD	6	100%	20	150	54	µg/kg	gamma distribution (0.05)	approximate gamma UCL	110
4,4′-DDE	6	100%	79	185	120	µg/kg	normal (0.05)	Student's-t UCL	160
4,4′-DDT	6	83%	3.15	245	59	µg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	220
Sum DDD	6	100%	31.8	190	67	µg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	140
Sum DDE	6	100%	79.5	188	120	µg/kg	normal (0.05)	Student's-t UCL	160

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Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Sum DDT	6	83%	3.5	295	73	µg/kg	gamma distribution (0.05)	approximate gamma UCL	270
Total DDTs	6	100%	152	673	263	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	624
Endrin	6	0%	0.5	15.5	5.3	µg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	17
Endrin aldehyde	6	0%	1	4.25	2.2	µg/kg	lognormal (0.05)	Student's-t UCL	3.1
Endrin ketone	6	0%	0.5	3.75	2.0	µg/kg	normal (0.05)	Student's-t UCL	2.9
Northern Pikeminnow									
Antimony	6	0%	0.0005	0.001	0.00067	mg/kg	non-parametric (0.05)	Mod-t UCL (adjusted for skewness)	0.00089
Arsenic	6	100%	0.19	0.36	0.26	mg/kg	gamma distribution (0.05)	approximate gamma UCL	0.34
Cadmium	6	100%	0.007	0.012	0.0098	mg/kg	normal (0.05)	Student's-t UCL	0.011
Chromium	6	83%	0.04	0.67	0.34	mg/kg	normal (0.05)	Student's-t UCL	0.58
Copper	6	100%	0.575	0.89	0.66	mg/kg	non-parametric (0.05)	Student's-t UCL	0.76
Lead	6	67%	0.0025	0.016	0.0088	mg/kg	normal (0.05)	Student's-t UCL	0.013
Mercury	6	100%	0.146	0.494	0.28	mg/kg	normal (0.05)	Student's-t UCL	0.4
Selenium	6	33%	0.15	0.4	0.22	mg/kg	non-parametric (0.05)	Student's-t UCL	0.31
Thallium	6	83%	0.0006	0.004	0.0023	mg/kg	normal (0.05)	Student's-t UCL	0.0033
Zinc	6	100%	16.4	20	18	mg/kg	normal (0.05)	Student's-t UCL	19
Total PCBs	6	100%	378	1,930	884	µg/kg	normal (0.05)	Student's-t UCL	1350
Aldrin	6	0%	2	6.5	4.2	µg/kg	normal (0.05)	Student's-t UCL	5.7
2,4′-DDD	6	17%	2.25	29	8.0	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	26
2,4′-DDE	6	17%	2	53	12	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	94
2,4′-DDT	6	17%	3.45	91	19	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	160
4,4′-DDD	6	83%	3.05	48	33	µg/kg	normal (0.05)	Student's-t UCL	48
4,4′-DDE	6	100%	82	545	250	µg/kg	normal (0.05)	Student's-t UCL	390
4,4′-DDT	6	17%	3.15	53	11	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	94
Sum DDD	6	83%	4.3	72	40	µg/kg	normal (0.05)	Student's-t UCL	60
Sum DDE	6	100%	86.9	598	264	µg/kg	normal (0.05)	Student's-t UCL	412

BZTO104(e)031009

Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Sum DDT	6	33%	3.5	94.2	29	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	190
Total DDTs	6	100%	163	764	335	µg/kg	gamma distribution (0.05)	approximate gamma UCL	593
Endrin	6	0%	2	9.5	4.7	µg/kg	normal (0.05)	Student's-t UCL	7
Endrin aldehyde	6	0%	2	4.9	3.6	µg/kg	non-parametric (0.05)	Student's-t UCL	4.7
Endrin ketone	6	0%	2	4.9	3.4	µg/kg	normal (0.05)	Student's-t UCL	4.3
Peamouth									
Antimony	4	0%	0.0015	0.0025	0.0020	mg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	0.0032
Arsenic	4	100%	0.35	0.48	0.43	mg/kg	normal (0.05)	Student's-t UCL	0.49
Cadmium	4	100%	0.021	0.053	0.032	mg/kg	normal (0.05)	Student's-t UCL	0.049
Chromium	4	75%	0.045	0.49	0.24	mg/kg	normal (0.05)	Student's-t UCL	0.45
Copper	4	100%	0.73	1.61	1.2	mg/kg	normal (0.05)	Student's-t UCL	1.7
Lead	4	100%	0.031	10.6	2.7	mg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	37
Mercury	4	100%	0.031	0.054	0.038	mg/kg	normal (0.05)	Student's-t UCL	0.051
Selenium	4	100%	0.3	0.4	0.35	mg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	0.45
Thallium	4	100%	0.0073	0.0093	0.0083	mg/kg	normal (0.05)	Student's-t UCL	0.0096
Zinc	4	100%	23.1	25.2	24.1	mg/kg	normal (0.05)	Student's-t UCL	25.3
Total PCBs	4	100%	140	300	193	µg/kg	normal (0.05)	Student's-t UCL	279
Aldrin	4	0%	0.5	0.85	0.61	µg/kg	normal (0.05)	Student's-t UCL	0.81
2,4′-DDD	4	0%	2	4.3	2.8	µg/kg	normal (0.05)	Student's-t UCL	4.1
2,4′-DDE	4	0%	0.5	1.05	0.73	µg/kg	normal (0.05)	Student's-t UCL	1.0
2,4′-DDT	4	25%	3.45	5.7	4.0	µg/kg	non-parametric (0.05)	Student's-t UCL	5.3
4,4′-DDD	4	100%	11.1	29.5	23	µg/kg	normal (0.05)	Student's-t UCL	32
4,4′-DDE	4	100%	109	185	130	µg/kg	normal (0.05)	Student's-t UCL	170
4,4′-DDT	4	50%	3.15	7	4.9	µg/kg	normal (0.05)	Student's-t UCL	7.4
Sum DDD	4	100%	13.1	32.4	25.4	µg/kg	normal (0.05)	Student's-t UCL	35.3
Sum DDE	4	100%	110	186	133	µg/kg	normal (0.05)	Student's-t UCL	175.5

BZTO104(e)031010

Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Sum DDT	4	50%	3.5	12.1	7.4	µg/kg	normal (0.05)	Student's-t UCL	13
Total DDTs	4	100%	135	225	167	µg/kg	normal (0.05)	Student's-t UCL	214.1
Endrin	4	0%	0.5	0.5	0.50	µg/kg	no distinct values in dataset[c]	maximum value	1.0
Endrin aldehyde	4	0%	0.5	0.8	0.59	µg/kg	gamma distribution (0.05)	approximate gamma UCL	0.84
Endrin ketone	4	0%	0.5	0.5	0.50	µg/kg	no distinct values in dataset[c]	maximum value	1.0
Sculpin									
Antimony	26	0%	0.0005	0.0045	0.00094	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.0017
Arsenic	26	100%	0.13	0.3	0.20	mg/kg	normal (0.05)	Student's-t UCL	0.22
Cadmium	26	100%	0.003	0.022	0.0093	mg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	0.011
Chromium	26	38%	0.035	0.35	0.098	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.18
Copper	26	100%	0.929	1.68	1.2	mg/kg	normal (0.05)	Student's-t UCL	1.3
Lead	26	96%	0.0149	0.96	0.14	mg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	0.2
Mercury	27	100%	0.025	0.086	0.042	mg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	0.047
Selenium	26	15%	0.1	0.3	0.16	mg/kg	non-parametric (0.05)	Student's-t UCL	0.18
Thallium	26	96%	0.0009	0.012	0.0043	mg/kg	gamma distribution (0.05)	approximate gamma UCL	0.0051
Zinc	26	100%	13.6	18	15	mg/kg	normal (0.05)	Student's-t UCL	16
Benzo(a)pyrene	26	0%	13.5	16.5	15	µg/kg	normal (0.05)	Student's-t UCL	16
Naphthalene	26	4%	13.5	42	16	µg/kg	non-parametric (0.05)	Student's-t UCL	18
Total PAHs	26	38%	14	342	120	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	380
Bis(2-ethylhexyl) phthalate	26	12%	41	28,000	1,500	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	13000
Dibutyl phthalate	26	0%	135	165	150	µg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	160
Total PCBs	26	100%	100	3,400	600	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	1300
PCB TEQ (birds)	9	100%	7.9	117	33.2	pg/g	lognormal (0.05)	95% H-UCL	97
PCB TEQ (mammals)	9	100%	4.82	43.7	14.8	pg/g	gamma distribution (0.05)	approximate gamma UCL	24.2
PCB TEQ (mammals 2006)	9	100%	2.7	24.1	7.97	pg/g	gamma distribution (0.05)	approximate gamma UCL	12.9
Dioxin TEQ (birds)	9	100%	2.34	166	24.2	pg/g	non-parametric (0.05)	99% Chebyshev UCL	202

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Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Dioxin TEQ (mammals)	9	100%	0.772	38.6	6.31	pg/g	assuming gamma distribution (0.05)	approximate gamma UCL	16.8
Dioxin TEQ (mammals 2006)	9	100%	0.701	31.8	5.33	pg/g	assuming gamma distribution (0.05)	approximate gamma UCL	13.7
Aldrin	26	0%	0.5	6.5	1.6	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	3.1
2,4′-DDD	26	15%	0.5	71	6.4	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	35
2,4′-DDE	26	12%	0.5	27	3.3	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	14
2,4′-DDT	26	65%	2.15	325	27	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	150
4,4′-DDD	26	73%	2	305	26	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	150
4,4′-DDE	26	73%	5.5	630	56	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	300
4,4′-DDT	26	81%	3.15	1,700	100	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	750
Sum DDD	26	73%	2.0	376	32	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	190
Sum DDE	26	73%	5.5	657	59	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	320
Sum DDT	26	85%	3.5	2,030	130	µg/kg	lognormal (0.05)	95% H-UCL	240
Total DDTs	26	100%	31.7	3,060	220	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	1400
Endrin	26	0%	0.5	15.5	3.2	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	13
Endrin aldehyde	26	0%	0.5	4.8	1.7	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	2.9
Endrin ketone	26	0%	0.5	4.8	1.3	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	2.2
beta-Hexachlorocyclohexane	26	46%	0.5	6.2	3.6	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	4.8
Smallmouth Bass									
Antimony	14	14%	0.0005	0.002	0.001	mg/kg	non-parametric (0.05)	Mod-t UCL (adjusted for skewness)	0.0012
Arsenic	14	100%	0.17	0.39	0.26	mg/kg	normal (0.05)	Student's-t UCL	0.29
Cadmium	14	71%	0.001	0.024	0.0055	mg/kg	gamma distribution (0.05)	approximate gamma UCL	0.0091
Chromium	14	86%	0.03	1.14	0.50	mg/kg	normal (0.05)	Student's-t UCL	0.68
Copper	14	100%	0.365	1.29	0.65	mg/kg	normal (0.05)	Student's-t UCL	0.78
Lead	14	100%	0.0048	0.303	0.042	mg/kg	non-parametric (0.05)	99% Chebyshev UCL	0.26
Mercury	14	100%	0.052	0.114	0.083	mg/kg	normal (0.05)	Student's-t UCL	0.092
Selenium	14	21%	0.1	0.4	0.19	mg/kg	non-parametric (0.05)	Student's-t UCL	0.23

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Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Thallium	14	100%	0.002	0.0085	0.0043	mg/kg	normal (0.05)	Student's-t UCL	0.0052
Zinc	14	100%	13.4	16.3	15	mg/kg	normal (0.05)	Student's-t UCL	15
Benzo(a)pyrene	14	0%	12.5	16.5	15	µg/kg	non-parametric (0.05)	Student's-t UCL	16
Naphthalene	14	14%	15	86	23	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	47
Total PAHs	14	50%	15	523	190	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	680
Bis(2-ethylhexyl) phthalate	14	14%	43	87,000	8,600	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	73000
Dibutyl phthalate	14	0%	125	165	150	µg/kg	non-parametric (0.05)	Student's-t UCL	160
Total PCBs	14	100%	109	4,950	1,200	µg/kg	gamma distribution (0.05)	approximate gamma UCL	1960
PCB TEQ (birds)	14	100%	15	82.7	32.0	pg/g	gamma distribution (0.05)	approximate gamma UCL	42
PCB TEQ (mammals)	14	100%	7.92	33.9	15.7	pg/g	gamma distribution (0.05)	approximate gamma UCL	20
PCB TEQ (mammals 2006)	14	100%	5.12	14	8.41	pg/g	assuming gamma distribution (0.05)	approximate gamma UCL	10.1
Dioxin TEQ (birds)	14	100%	2.26	22.5	8.18	pg/g	lognormal (0.05)	95% H-UCL	12
Dioxin TEQ (mammals)	14	100%	1.45	10.4	4.07	pg/g	gamma distribution (0.05)	approximate gamma UCL	5.5
Dioxin TEQ (mammals 2006)	14	100%	1.29	7.77	3.47	pg/g	assuming gamma distribution (0.05)	approximate gamma UCL	4.56
Aldrin	14	0%	0.5	6.5	2.5	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	4.9
2,4′-DDD	14	21%	2.45	29	6.6	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	15
2,4′-DDE	14	0%	0.5	3.75	2.4	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	3.9
2,4′-DDT	14	14%	3.45	30	6.3	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	15
4,4′-DDD	14	100%	11	110	38	µg/kg	gamma distribution (0.05)	approximate gamma UCL	51
4,4′-DDE	14	100%	53	220	130	µg/kg	normal (0.05)	Student's-t UCL	150
4,4′-DDT	14	64%	3.15	130	29	µg/kg	gamma distribution (0.05)	approximate gamma UCL	55
Sum DDD	14	100%	15.3	139	44	µg/kg	gamma distribution (0.05)	approximate gamma UCL	60
Sum DDE	14	100%	56.8	222	130	µg/kg	normal (0.05)	Student's-t UCL	160
Sum DDT	14	64%	3.5	160	34	µg/kg	gamma distribution (0.05)	approximate gamma UCL	66
Total DDTs	14	100%	79.7	416	214	µg/kg	normal (0.05)	Student's-t UCL	262
Endrin	14	0%	0.5	15.5	5.0	µg/kg	lognormal (0.05)	95% Chebyshev (MVUE) UCL	12

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Table 6-2. EPCs in Invertebrates and Fish

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Data Distribution	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Endrin aldehyde	14	0%	0.5	10	3.1	µg/kg	lognormal (0.05)	95% H-UCL	5.6
Endrin ketone	14	0%	0.5	10	2.9	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	6.6

[a] The concentration value was represented by the detected concentration or one-half the detection limit for non-detected values.

[b] EPC equals ProUCL-recommended value or maximum.

[c] No statistic was calculated by ProUCL where all the data values in a dataset were the same (i.e., no unique values). Where no statistic was calculated, the maximum value was used to represent the EPC.

EPC – exposure point concentration

MVUE – minimum variance unbiased estimator

PAH – polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

UCL – upper confidence limit

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Table 6-3. EPCs in Shorebird Tissue and Sediment

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Distribution Type	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Clam									
Arsenic	26	100	0.802	1.25	0.94	mg/kg	gamma distribution (0.05)	approximate gamma UCL	0.97
Cadmium	26	100	0.05	0.218	0.094	mg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	0.11
Chromium	26	100	0.4	0.94	0.65	mg/kg	normal (0.05)	Student's-t UCL	0.69
Lead	26	100	0.024	0.316	0.1	mg/kg	lognormal (0.05)	95% H-UCL	0.13
Mercury	24	100	0.006	0.013	0.009	mg/kg	normal (0.05)	Student's-t UCL	0.01
Selenium	26	92	0.05	0.21	0.11	mg/kg	normal (0.05)	Student's-t UCL	0.13
Thallium	3	100	0.00035	0.0007	5E-04	mg/kg	insufficient data available (n $\leq$ 3)	maximum value	0.0007
Zinc	26	100	21.4	48	34	mg/kg	normal (0.05)	Student's-t UCL	37
Tributyltin ion	22	82	0.9	63	9.1	µg/kg	lognormal (0.05)	95% H-UCL	15
Benzo(a)pyrene	28	89	1.4	490	27	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	200
Naphthalene	28	7	0.485	33	3.6	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	18
Total PAHs	28	100	34.8	4980	483	µg/kg	lognormal (0.05)	95% H-UCL	699
Bis(2-ethylhexyl) phthalate	27	7	26.5	170	50	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	82
Dibutyl phthalate	27	4	8	1300	83	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	570
Total PCBs	30	100	50.1	2660	250	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	622
PCB TEQ (birds)	27	100	3.46	71.8	13.6	pg/g	non-parametric (0.05)	95% Chebyshev UCL	26
PCB TEQ (mammals)	27	100	0.761	12	2.79	pg/g	lognormal (0.05)	95% H-UCL	3.57
PCB TEQ (mammals 2006)	27	100	0.534	8.56	1.9	pg/g	assuming gamma distribution (0.05)	approximate gamma UCL	2.4
Dioxin TEQ (birds)	26	100	0.75	32.1	4.42	pg/g	non-parametric (0.05)	95% Chebyshev UCL	10.5
Dioxin TEQ (mammals)	26	100	0.432	6.94	1.47	pg/g	non-parametric (0.05)	95% Chebyshev UCL	2.86
Dioxin TEQ (mammals 2006)	26	100	0.407	5.63	1.28	pg/g	non-parametric (0.05)	95% Chebyshev UCL	2.39
Aldrin	30	90	0.126	5.07	0.43	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	1.1
2,4′-DDD	30	100	0.394	82	8	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	39
2,4′-DDE	30	97	0.0978	12.5	1.3	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	5.8
2,4′-DDT	30	97	0.167	39	3.3	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	18
4,4′-DDD	30	100	1.76	160	22	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	92
4,4′-DDE	30	100	4.79	94.5	19	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	35
4,4′-DDT	30	97	0.611	75	7.6	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	37

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Table 6-3. EPCs in Shorebird Tissue and Sediment

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Distribution Type	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Sum DDD	30	100	2.15	242	30	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	130
Sum DDE	30	100	4.89	107	20.1	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	37.7
Sum DDT	30	97	0.778	114	11	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	55
Total DDTs	30	100	7.82	463	60.5	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	139
Endrin	30	53	0.00201	1.2	0.086	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	0.53
Endrin aldehyde	30	3	0.0075	0.5	0.06	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	0.32
Endrin ketone	30	33	0.00078	0.5	0.066	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	0.35
Laboratory Worm									
Arsenic	29	100	0.285	3.04	1.3	mg/kg	normal (0.05)	Student's-t UCL	1.4
Cadmium	29	100	0.0364	0.254	0.068	mg/kg	non-parametric (0.05)	Student's-t UCL	0.084
Chromium	29	100	0.3	0.82	0.52	mg/kg	gamma distribution (0.05)	approximate gamma UCL	0.57
Copper	29	100	1.83	3.49	2.4	mg/kg	gamma distribution (0.05)	approximate gamma UCL	2.5
Lead	29	100	0.167	0.847	0.35	mg/kg	lognormal (0.05)	Student's-t UCL	0.4
Mercury	28	100	0.003	0.0105	0.006	mg/kg	assuming gamma distribution (0.05)	approximate gamma UCL	0.0062
Selenium	29	100	0.08	0.37	0.2	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.29
Zinc	29	100	18.2	31.5	26	mg/kg	normal (0.05)	Student's-t UCL	27
Tributyltin ion	29	38	0.22	49	5.1	µg/kg	lognormal (0.05)	95% H-UCL	9.6
Benzo(a)pyrene	29	100	2.2	970	70	µg/kg	lognormal (0.05)	95% H-UCL	110
Naphthalene	29	17	0.23	31	2.4	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	13
Total PAHs	29	100	83.6	23300	2320	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	12000
Bis(2-ethylhexyl) phthalate	29	52	26.5	200	85	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	130
Dibutyl phthalate	29	14	8	470	80	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	260
Total PCBs	29	100	44.8	4310	669	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	2790
PCB TEQ (birds)	29	100	3.81	732	66.5	pg/g	non-parametric (0.05)	99% Chebyshev UCL	371
PCB TEQ (mammals)	29	100	0.935	49.1	6.86	pg/g	non-parametric (0.05)	99% Chebyshev UCL	26.8
PCB TEQ (mammals 2006)	29	100	0.676	32.4	4.63	pg/g	non-parametric (0.05)	99% Chebyshev UCL	17.6
Dioxin TEQ (birds)	29	100	1.75	932	43.5	pg/g	non-parametric (0.05)	99% Chebyshev UCL	362
Dioxin TEQ (mammals)	29	100	1.1	472	21.1	pg/g	non-parametric (0.05)	99% Chebyshev UCL	182
Dioxin TEQ (mammals 2006)	29	100	1.02	448	19.5	pg/g	non-parametric (0.05)	99% Chebyshev UCL	172

BZTO104(e)031016

Table 6-3. EPCs in Shorebird Tissue and Sediment

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Distribution Type	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Aldrin	29	97	0.043	37	1.7	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	14
2,4′-DDD	29	100	0.611	204	17	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	110
2,4′-DDE	29	100	0.127	52.7	3.3	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	22
2,4′-DDT	29	66	0.00955	3.13	0.28	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	1.5
4,4′-DDD	29	100	4.08	1060	80	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	500
4,4′-DDE	29	100	9.52	171	29	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	59
4,4′-DDT	29	97	0.0456	37.9	2.9	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	19
Sum DDD	29	100	4.78	1260	97.8	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	611
Sum DDE	29	100	9.66	188	32	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	66
Sum DDT	29	100	0.0562	40	3.19	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	20.7
Total DDTs	29	100	14.5	1490	133	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	725
Endrin	29	28	0.001855	0.157	0.017	µg/kg	lognormal (0.05)	95% H-UCL	0.024
Endrin aldehyde	29	0	0.000346	0.0192	0.009	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.012
Endrin ketone	29	10	0.00073	0.04	0.02	µg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.027
Shorebird sediment									
Arsenic	39	100	0.7	8	2.8	mg/kg	gamma distribution (0.05)	approximate gamma UCL	3.1
Cadmium	39	100	0.03	4.21	0.27	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.74
Chromium	39	100	11	83.6	23	mg/kg	non-parametric (0.05)	Student's-t UCL	26
Copper	39	100	8.8	194	32	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	57
Lead	39	100	4.7	242	24	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	52
Mercury	39	59	0.0035	0.412	0.040	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.088
Selenium	39	15	0.02	0.15	0.063	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.09
Zinc	39	100	52.7	1230	130	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	260
Benzo(a)pyrene	39	95	0.12	1300	110	µg/kg	lognormal (0.05)	95% Chebyshev (MVUE) UCL	330
Naphthalene	39	28	0.365	41	9.2	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	26
Total PAHs	39	100	6.685	11865	1,010	µg/kg	lognormal (0.05)	95% Chebyshev (MVUE) UCL	2210
Bis(2-ethylhexyl) phthalate	39	46	1	2600	100	µg/kg	lognormal (0.05)	95% Chebyshev (MVUE) UCL	160
Dibutyl phthalate	39	54	1.45	690	26	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	200
Total PCBs	39	56	0.95	1770	86	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	540

BZTO104(e)031017

Table 6-3. EPCs in Shorebird Tissue and Sediment

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Distribution Type	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
PCB TEQ (birds)	12	100	0.153	549	56.1	pg/g	assuming gamma distribution (0.05)	adjusted gamma UCL	250
PCB TEQ (mammals)	12	100	0.0289	40.2	4.66	pg/g	gamma distribution (0.05)	adjusted gamma UCL	17.1
PCB TEQ (mammals 2006)	12	100	0.0272	31.3	3.68	pg/g	gamma distribution (0.05)	adjusted gamma UCL	13.2
Dioxin TEQ (birds)	23	100	0.0568	248	13.7	pg/g	lognormal (0.05)	97.5% Chebyshev (MVUE) UCL	31.7
Dioxin TEQ (mammals)	23	100	0.0731	75.8	5.64	pg/g	lognormal (0.05)	95% Chebyshev (MVUE) UCL	13.2
Dioxin TEQ (mammals 2006)	23	100	0.0731	62.1	4.87	pg/g	lognormal (0.05)	95% Chebyshev (MVUE) UCL	11.6
Aldrin	39	10	0.0132	2.48	0.16	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	0.84
2,4′-DDD	39	46	0.0136	89	4.2	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	28
2,4′-DDE	39	10	0.0144	26	1.3	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	8.7
2,4′-DDT	39	44	0.01905	123	5.8	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	39
4,4′-DDD	39	64	0.0205	81.5	5.2	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	29
4,4′-DDE	39	49	0.0192	122	6.9	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	44
4,4′-DDT	39	69	0.0239	184	11	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	67
Sum DDD	39	64	0.0205	129	9.3	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	53
Sum DDE	39	49	0.0192	135	8.1	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	52
Sum DDT	39	72	0.19	307	17	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	110
Total DDTs	39	74	0.195	490.3	34	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	200
Endrin	37	14	0.01675	1.92	0.19	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	0.75
Endrin aldehyde	39	0	0.0184	0.6	0.11	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	0.3
Endrin ketone	39	8	0.0124	0.55	0.13	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	0.35

[a] The concentration value was represented by the detected concentration or one-half the detection limit for non-detected values.
[b] EPC equals ProUCL-recommended value or maximum.

EPC – exposure point concentration
MVUE – minimum variance unbiased estimator
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
UCL – upper confidence limit

BZTO104(e)031018

Table 6-4. EPCs in Surface Sediment

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Distribution Type	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Site-wide sediment									
Antimony	1027	100%	0.015	32.1	0.97	mg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	1.4
Arsenic	1152	89%	0.97	75.6	4.5	mg/kg	non-parametric (0.05)	Student's-t UCL	4.8
Cadmium	1134	93%	0.000795	46.2	0.41	mg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	0.69
Chromium	1139	99%	4.07	774	33	mg/kg	non-parametric (0.05)	Student's-t UCL	35
Copper	1139	100%	6.19	1,080	57	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	67
Lead	1139	100%	2.5	1,950	37	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	52
Mercury	1131	94%	0.0035	4.84	0.10	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.12
Selenium	1058	46%	0.015	20	1.6	mg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	2.3
Thallium	189	87%	0.026	27	7.9	mg/kg	non-parametric (0.05)	99% Chebyshev UCL	13
Zinc	1139	100%	33	2,850	150	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	180
Tributyltin ion	250	94%	0.115	47,000	590	µg/kg	lognormal (0.05)	95% H-UCL	460
Benzo(a)pyrene	1187	97%	0.12	340,000	2,000	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	4,600
Naphthalene	1188	65%	0.215	73,000	430	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	1,200
Total PAHs	1187	100%	5.0	7,260,000	31,000	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	78,000
Bis(2-ethylhexyl) phthalate	1130	58%	1	440,000	880	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	3,300
Dibutyl phthalate	1129	32%	1.45	3,790	35	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	63
Total PCBs	884	76%	0.85	30,800	170	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	400
PCB TEQ (birds)	161	100%	0.153	5,610	76.7	pg/g	non-parametric (0.05)	97.5% Chebyshev UCL	308
PCB TEQ (mammals)	161	100%	0.0289	324	9.40	pg/g	non-parametric (0.05)	97.5% Chebyshev UCL	25.4
PCB TEQ (mammals 2006)	161	100%	0.0272	239	7.15	pg/g	non-parametric (0.05)	97.5% Chebyshev UCL	19.1
Dioxin TEQ (birds)	156	100%	0.0568	35,300	254	pg/g	non-parametric (0.05)	99% Chebyshev UCL	2,510
Dioxin TEQ (mammals)	156	100%	0.029	16,600	121	pg/g	lognormal (0.05)	95% H-UCL	30.5
Dioxin TEQ (mammals 2006)	156	100%	0.0285	14,100	103	pg/g	lognormal (0.05)	95% H-UCL	26.6
Aldrin	816	26%	0.00333	691	2.3	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	7.8
2,4′-DDD	854	69%	0.0136	710	9.5	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	18
2,4′-DDE	853	29%	0.00495	1,000	3.6	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	16
2,4′-DDT	854	54%	0.01105	2,000	12	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	36
4,4′-DDD	910	84%	0.0205	2,780	27	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	57
4,4′-DDE	907	82%	0.0192	2,240	13	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	33

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Table 6-4. EPCs in Surface Sediment

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Distribution Type	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
4,4′-DDT	895	73%	0.0239	12,000	84	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	300
Sum DDD	910	85%	0.021	3,040	36	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	72
Sum DDE	907	83%	0.0192	2,530	16	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	40
Sum DDT	909	79%	0.0221	12,500	94	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	330
Total DDTs	910	90%	0.027	16,200	150	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	340
Endrin	614	12%	0.00198	100	2.2	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	4.6
Endrin aldehyde	864	4%	0.0184	100	1.2	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	2.9
Endrin ketone	864	18%	0.00104	100	1.5	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	3.3
0-20 ft depth sediment									
Antimony	602	77%	0.015	32.1	1.1	mg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	1.8
Arsenic	678	89%	0.97	75.6	4.9	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	5.9
Cadmium	664	88%	0.000795	46.2	0.44	mg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	0.9
Chromium	667	100%	4.07	774	35	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	42
Copper	667	100%	6.19	772	58	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	72
Lead	667	100%	2.65	1,950	45	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	67
Mercury	661	93%	0.0035	2.01	0.10	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	0.13
Selenium	645	41%	0.015	17.5	1.5	mg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	2.5
Thallium	121	84%	0.026	27	7.3	mg/kg	non-parametric (0.05)	99% Chebyshev UCL	14
Zinc	667	100%	33	2,850	170	mg/kg	non-parametric (0.05)	95% Chebyshev UCL	200
Tributyltin ion	151	94%	0.115	47,000	760	µg/kg	lognormal (0.05)	95% H-UCL	340
Benzo(a)pyrene	702	96%	0.12	160,000	1,700	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	4,100
Naphthalene	702	66%	0.215	73,000	500	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	2,300
Total PAHs	702	100%	6.32	2,450,000	26,000	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	66,000
Bis(2-ethylhexyl) phthalate	663	56%	1	440,000	1,300	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	5500
Dibutyl phthalate	663	29%	1.45	3,790	45	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	91
Total PCBs	491	77%	0.85	30,800	230	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	640
PCB TEQ (birds)	122	100%	0.153	5,610	96.4	pg/g	non-parametric (0.05)	97.5% Chebyshev UCL	401
PCB TEQ (mammals)	122	100%	0.0289	324	11.4	pg/g	non-parametric (0.05)	97.5% Chebyshev UCL	32.4
PCB TEQ (mammals 2006)	122	100%	0.0272	239	8.69	pg/g	non-parametric (0.05)	97.5% Chebyshev UCL	24.3
Dioxin TEQ (birds)	116	100%	0.0568	35,300	333	pg/g	non-parametric (0.05)	99% Chebyshev UCL	3360

BZTO104(e)031020

Table 6-4. EPCs in Surface Sediment

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Distribution Type	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Dioxin TEQ (mammals)	116	100%	0.029	16,600	158	pg/g	non-parametric (0.05)	99% Chebyshev UCL	1,580
Dioxin TEQ (mammals 2006)	116	100%	0.0285	14,100	135	pg/g	non-parametric (0.05)	99% Chebyshev UCL	1,340
Aldrin	519	24%	0.00333	691	3.2	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	12
2,4′-DDD	531	65%	0.0136	710	13	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	27
2,4′-DDE	530	30%	0.00495	1,000	5.4	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	26
2,4′-DDT	531	50%	0.01105	2,000	19	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	78
4,4′-DDD	568	80%	0.0205	2,780	37	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	110
4,4′-DDE	565	75%	0.0192	2,240	19	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	50
4,4′-DDT	559	70%	0.0239	12,000	130	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	470
Sum DDD	568	82%	0.021	3,040	49	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	110
Sum DDE	565	76%	0.0192	2,530	23	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	61
Sum DDT	567	74%	0.0221	12,500	140	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	520
Total DDTs	568	86%	0.0307	16,200	220	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	700
Endrin	402	14%	0.00198	100	2.9	µg/kg	non-parametric (0.05)	99% Chebyshev UCL	8.5
Endrin aldehyde	547	4%	0.0184	100	1.7	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	4.2
Endrin ketone	547	17%	0.00104	100	2.0	µg/kg	non-parametric (0.05)	97.5% Chebyshev UCL	4.7

[a] The concentration value was represented by the detected concentration or one-half the detection limit for non-detected values.

[b] EPC equals ProUCL-recommended value or maximum.

EPC – exposure point concentration

PAH – polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

UCL – upper confidence limit

BZTO104(e)031021

Table 6-5. EPCs in Surface Water

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Distribution Type	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
Benthic invertebrates – Near bottom samples									
Peristaltic									
Zinc (dissolved)	56	88	0.000585	0.0419	0.00264	mg/L	non-parametric (0.05)	95% Chebyshev UCL	0.0058
Benzo(a)anthracene	56	18	0.00105	0.11	0.00491	µg/L	non-parametric (0.05)	95% Chebyshev UCL	0.013
Benzo(a)pyrene	56	12	0.0008	0.15	0.00489	µg/L	non-parametric (0.05)	95% Chebyshev UCL	0.016
4-Chloro-3-methylphenol	55	11	0.0145	0.65	0.0576	µg/L	non-parametric (0.05)	95% Chebyshev UCL	0.12
Total PCBs	38	13	0.00125	0.01618	0.00252	µg/L	non-parametric (0.05)	95% Chebyshev UCL	0.005
2,4′-DDD	43	0	0.000236	0.000269	0.000248	µg/L	non-parametric (0.05)	Student's-t UCL	0.00025
2,4′-DDT	43	2	0.000236	0.0187	0.000678	µg/L	non-parametric (0.05)	95% Chebyshev UCL	0.0025
4,4′-DDD	43	2	0.000236	0.000839	0.000262	µg/L	non-parametric (0.05)	Student's-t UCL	0.00029
4,4′-DDT	43	5	0.000236	0.00115	0.000280	µg/L	non-parametric (0.05)	Student's-t UCL	0.00032
Total DDTs	43	9	0.000236	0.0199	0.000836	µg/L	non-parametric (0.05)	95% Chebyshev UCL	0.0028
XAD									
Benzo(a)anthracene	15	100	0.000118	0.00613	0.00144	µg/L	gamma distribution (0.05)	approximate gamma UCL	0.00244
Benzo(a)pyrene	15	93	0.0000183	0.00664	0.00130	µg/L	gamma distribution (0.05)	approximate gamma UCL	0.00261
Total PCBs	15	100	0.000379	0.0120	0.00256	µg/L	gamma distribution (0.05)	approximate gamma UCL	0.00428
2,4′-DDD	15	100	0.00000566	0.002057	0.000413	µg/L	non-parametric (0.05)	99% Chebyshev UCL	0.0021
2,4′-DDT	15	93	2.29E-07	0.000525	8.76E-05	µg/L	lognormal (0.05)	99% Chebyshev (MVUE) UCL	0.001
4,4′-DDD	15	100	0.0000120	0.00325	0.000842	µg/L	non-parametric (0.05)	99% Chebyshev UCL	0.004
4,4′-DDT	15	100	0.00000134	0.003862	0.000552	µg/L	lognormal (0.05)	99% Chebyshev (MVUE) UCL	0.00591
Total DDTs	15	100	0.0000496	0.00976	0.00209	µg/L	non-parametric (0.05)	99% Chebyshev UCL	0.0102
Fish – All surface water samples									
Peristaltic									
Zinc (dissolved)	74	91	0.000585	0.0419	0.0025	mg/L	non-parametric (0.05)	95% Chebyshev UCL	0.0049
Benzo(a)anthracene	74	16	0.00105	0.11	0.0044	µg/L	non-parametric (0.05)	95% Chebyshev UCL	0.011
Benzo(a)pyrene	74	9	0.0008	0.15	0.0041	µg/L	non-parametric (0.05)	95% Chebyshev UCL	0.013

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Table 6-5. EPCs in Surface Water

Analyte	N	Detection Frequency	Concentration Value[a]			Unit	Distribution Type	Statistic Used	EPC[b]
			Minimum	Maximum	Mean				
4-Chloro-3-methylphenol	73	10	0.0145	0.65	0.052	µg/L	non-parametric (0.05)	95% Chebyshev UCL	0.1
Total PCBs	47	13	0.00125	0.018	0.0026	µg/L	non-parametric (0.05)	95% Chebyshev UCL	0.0051
2,4′-DDD	52	0	0.000236	0.000269	0.00025	µg/L	non-parametric (0.05)	Student's-t UCL	0.00025
2,4′-DDT	52	2	0.000236	0.0187	0.00060	µg/L	non-parametric (0.05)	95% Chebyshev UCL	0.0021
4,4′-DDD	52	2	0.000236	0.000839	0.00026	µg/L	non-parametric (0.05)	Student's-t UCL	0.00028
4,4′-DDT	52	4	0.000236	0.00115	0.00027	µg/L	non-parametric (0.05)	Student's-t UCL	0.00031
Total DDTs	52	8	0.000236	0.0199	0.00073	µg/L	non-parametric (0.05)	95% Chebyshev UCL	0.0024
XAD									
Benzo(a)anthracene	24	100	0.000052	0.00613	0.0011	µg/L	gamma distribution (0.05)	approximate gamma UCL	0.0017
Benzo(a)pyrene	24	96	0.0000183	0.006636	0.000999	µg/L	gamma distribution (0.05)	approximate gamma UCL	0.00167
Total PCBs	24	100	0.000172	0.0120	0.00175	µg/L	data are lognormal (0.05)	95% Chebyshev (MVUE) UCL	0.00325
2,4′-DDD	24	100	0.00000326	0.002057	0.00027	µg/L	non-parametric (0.05)	99% Chebyshev UCL	0.0014
2,4′-DDT	24	96	2.285E-07	0.000525	5.6E-05	µg/L	non-parametric (0.05)	99% Chebyshev UCL	0.00031
4,4′-DDD	24	100	0.0000116	0.00325	0.00054	µg/L	non-parametric (0.05)	99% Chebyshev UCL	0.0027
4,4′-DDT	24	100	0.00000134	0.003862	0.000351	µg/L	non-parametric (0.05)	99% Chebyshev UCL	0.00212
Total DDTs	24	100	0.0000431	0.00976	0.00135	µg/L	non-parametric (0.05)	99% Chebyshev UCL	0.00672

[a] The concentration value was represented by the detected concentration or one-half the detection limit for non-detected values.
[b] EPC equals ProUCL-recommended value or maximum.

EPC – exposure point concentration
MVUE – minimum variance unbiased estimator
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
UCL – upper confidence limit

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APPENDIX G: ECOLOGICAL RISK ASSESSMENT

Attachment G2
Screening Assessment for Benthic Invertebrates

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1.0 INITIAL SCREENING PROCESS

In the initial screening process for the benthic community, all lines of evidence associated with the four measures of exposure and effects were evaluated (see Section 3.0 of Appendix G). These four measures were sediment toxicity testing and modeling, sediment chemistry comparison to sediment quality values (SQVs) in literature, benthic chemistry tissue data comparison to tissue residue toxicity reference values (TRVs), and water data comparison to ambient water quality criteria (AWQC) or ecological screening levels. No screening process was associated with benthic toxicity testing and modeling, and it is therefore presented in its entirety in the Section 3.0 of Appendix G. The remaining three measures of exposure and effects included an initial screening process, which is presented in the following sections: evaluation of sediment chemistry data using literature-based SQVs (Section 2.0), evaluation of tissue chemistry data (Section 3.0), and evaluation of water data (Section 7.0).

2.0 LITERATURE-BASED SQV ASSESSMENT

One line of evidence (LOE) considered was based on exposure and effect to the benthic community predicted by sediment SQVs available in literature. Literature-based SQVs are either consensus-based[1] or empirical[2] SQVs, and both were included in this line of evidence in the risk assessment of the benthic community. In the *Interpretive Report: Estimating Risks to Benthic Organisms Using Predictive Models Based on Sediment Toxicity Tests,* hereafter referred to as the Benthic Interpretive Report (Windward et al. 2006), several of the existing SQVs were evaluated for their reliability in predicting toxicity to benthic invertebrates based on the bioassays conducted and decision rules applied in the Study Area. The results of predicting toxicity using literature-based SQVs were compared to the outcomes based on the site-specific SQVs developed using models presented in the Benthic Interpretive Report. Because the SQV evaluation was an early screening step (conducted as part of the interpretive report (Windward et al. 2006) and was initiated before some details of the project were finalized, slightly different combinations of endpoints and data treatment methods were used compared to the predictive models (see Section 3.0 of Appendix G). However, the overall comparison of the reliability of published SQVs with site-specific SQVs is still valid. The following published SQVs were evaluated:

- **Threshold effects levels (TELs)/probable effects levels (PELs) –** TELs are intended to represent chemical concentrations below which biological effects rarely occur. PELs are intended to represent chemical concentrations above which adverse biological effects are likely occur. TEL/PEL values have been adopted in Canada and several states (CCME 2002).
- **Threshold effects concentrations (TECs)/probable effects concentrations (PECs) –** Consensus-based SQVs have been proposed by a group of private and agency sediment researchers in an attempt to unify the wide variety of SQVs available in the literature. The TECs/PECs (MacDonald et al. 2000) are similar in concept to TELs/PELs (Ingersoll et al. 2000).
- **Lowest effect levels (LELs)/severe effects levels (SELs) –** The screening level (SL) concentration approach was developed by the Ontario Ministry of the Environment and is based on the presence or absence of benthic species in freshwater sediments (Persaud et al. 1993). The LEL corresponds to a level at which effects would be expected in only 5% of benthic species, while the SEL represents a level at which effects would be expected in 95% of benthic species.

[1] Consensus-based SQVs were developed to "combine" the wide variety of SQVs available in the literature. SQVs with similar narrative intent were grouped together.
[2] Empirical SQVs were developed based on relationship between sediment chemistry and results of toxicity tests.

- **Washington State Freshwater Sediment Quality Standards (SQS)/cleaning screening levels (CSLs)** – The floating percentile model was developed to improve the reliability of freshwater SQVs for Washington State (Avocet and SAIC 2002; Avocet 2003). SQS and CSLs have been calculated (but not promulgated) and are currently applicable to freshwater sediments in Washington State (Avocet 2003).
- **Quotient methods** – Quotient methods were developed as an approach to increase the predictive ability of certain SQVs (Long et al. 1998) and have been applied to TELs/PELs and TECs/PECs (as described above). In the quotient methods the concentrations of individual or sums of chemicals are divided by their respective SQV and the mean of these concentration-to-SQV quotients are determined. Two quotient methods – one using sums and one using individual chemicals – were evaluated. One additional quotient method, the effects range – median (ERM) quotient, was evaluated at a later time per request of the US Environmental Protection Agency (EPA).

Seven parameters were used in the reliability evaluation (hits refers to the result based on the decision rule used in the source data):

- **False negative rates** – Incorrectly predicted no-hits/total hits
- **False positive rates** – Incorrectly predicted hits/total no-hits
- **Sensitivity** – Correctly predicted hits/total hits
- **Efficiency** – Correctly predicted no-hits/total no-hits
- **Predicted hit reliability** – Correctly predicted hits/total predicted hits
- **Predicted no-hit reliability** – Correctly predicted no-hits/total predicted no-hits
- **Overall reliability** – Correctly predicted stations/total stations

For each existing SQV set, the more protective of the two thresholds (i.e., TEC, TEL, LEL, and SQS) was compared to Effects Level 2[3] derived from the site-specific models and the higher of the two thresholds (i.e., PEC, PEL, SEL, and CSL) was compared to Effects Level 3 derived from site-specific data, consistent with the narrative intent of these SQVs (i.e., minor and moderate biological effects). The ERM quotient was compared to both effects levels. Each of the four individual toxicity test endpoints (survival and growth for the *Chironomus tentans* and *Hyalella azteca*

[3] Biological effects were defined to correspond conceptually to "minor effects level" (Level 2) and "moderate effects level" (Level 3). The two levels were based on an approach suggested by EPA, the National Oceanic and Atmospheric Administration, and the Oregon Department of Environmental Quality (ODEQ 1999) for this project (EPA 2005b) and were defined as 80 and 70% of the response observed in the control sediment (for further details, see the ERA Appendix, Section 3.2.1).

toxicity tests) was assessed. In addition, a pooled endpoint was derived by combining all four endpoints from the two tests.

This summary focuses on five of the reliability parameters: false negative rates, false positive rates, predicted no-hit reliability, predicted hit reliability, and overall reliability. The complete reliability assessment with all seven reliability parameters is presented in the Benthic Interpretive Report (Windward et al. 2006) and a more complete discussion of the reliability analysis is presented in Appendix A of that report. At the Effects Level 2 comparison, the TEC, TEL, and LEL values performed similarly and more conservatively (declared more impacts). In all three cases, the SQV sets had very low false negative rates (0 to 8%). However, these SQV sets classified nearly every sample as a hit, leading to very high false positive rates (80 to 99%). In general, these SQV sets predicted that all or nearly all samples would be hits, and the proportion of correctly predicted hits simply reflected the proportion of actual biological hits in the dataset. Therefore, these SQV sets are not really useful in making correct predictions at Effects Level 2. The predicted no-hit and predicted hit reliability parameters for the three SQV sets ranged from 67 to 100% and 9 to 59%, respectively.

The Washington State freshwater SQS and ERM quotient values were less conservative than the other three SQV sets. The false negative rate for these SQV sets ranged from 15 to 38%, and the false positive rate ranged from 36 to 74%. The predicted no-hit and the predicted hit reliability parameters for these two SQV sets ranged from 54 to 96% and 12 to 61%. At Effects Level 2 comparison, the overall reliability for all five SQV sets ranged from 9 to 66%.

At the Effects Level 3 comparison, most of the SQV sets appeared to perform better. The TEC, TEL, LEL, and CSL values had false negative rate ranged from 17 to 59% and the false positive rate ranged from 16 to 48%. The false negative rates for the ERM quotient were relatively low (16 to 22%); however the false positive rates varied a lot depending on which endpoint was evaluated (4 to 74%). Among all the SQV sets, there was a better balance between false negatives and false positives, although judging by the low predicted hit reliability values (9 to 58%), there was still a tendency to over-predict actual hits by a substantial amount (three times the actual number of hits). The predicted no-hit reliability for all the SQV sets ranged from 55 to 97% and the overall reliability from 37 to 78%.

The reliability of the two quotient methods, SQG-Q and PEL-Q, was graphically evaluated in the Benthic Interpretive Report (Windward et al. 2006). The graphs showed the full range of quotient values (SQG-Q and PEL-Q) and the percentage of hit/no-hit classification. Ideally, both the percentage of no-hits below the quotient value and the percentage of hits above the quotient value should be high (e.g., above 80%) in order for a selected quotient value to have good reliability in predicting both hits and no-hits. This did not occur at any effects levels throughout the range of possible quotient values, except in some cases at the extreme ends of the data

distribution. Setting values at the ends of the distributions would not be helpful because only a few stations fall below these levels (at the low end) or above these levels (at the high end).

In summary, the initial reliability screening process found that none of the SQV sets available in the literature performed well enough to use them in predicting biological effects at the Portland Harbor Superfund site, suggesting that development of a site-specific SQV set or predictive model was necessary to improve reliability in predicting biological effects to the benthic community.

The predicted benthic toxicity based on a comparison between the Study Area surface sediment chemistry dataset and each set of SQVs (i.e., TEC/PEC, TEL/PEL, LEL/SEL, SQS/CSL, and ERM quotient methods) are presented in Figures 2-1 through 2-5.

3.0 TISSUE-RESIDUE ASSESSMENT

Exposure and effect to the benthic community was assessed by comparing tissue residue concentrations to TRVs available in the literature. The tissue residue data included two different empirical methods, field-collected and laboratory-exposed organisms, which were presented as two lines of evidence. Field-collected tissue samples included clams (*Corbicula* spp.), crayfish, and invertebrates collected with multiplate samplers deployed in the water column; laboratory-exposed tissue samples included clams (*Corbicula fluminea*) and worms (*Lumbriculus variegatus*). All tissue types were used in the risk assessment for the benthic community. Field-collected and laboratory-exposed clam tissue samples were used in the risk assessment for clams; field-collected crayfish tissue samples were used in the risk assessment for crayfish.

This section presents the initial screening process for the tissue residue assessment, including the tissue-residue screening and chemical of potential concern (COPC) identification (Section 3.1). An evaluation of the predicted tissue line of evidence is presented in Section 3.2.

3.1 EMPIRICAL TISSUE-RESIDUE SCREEN AND COPC IDENTIFICATION

Chemicals included in the tissue assessment were first identified as chemicals of interest (COIs) (Section 3.1.1) if the chemicals were detected in tissue samples and then as Round 2 COPCs (Section 3.1.2) if the maximum concentrations exceeded the TRVs.

3.1.1 Selection of COIs

COIs for benthic invertebrates were defined as all chemicals detected in invertebrate tissue samples from the ERA dataset, excluding crustal elements (i.e., aluminum and manganese; see Section 2.0 of Appendix G). Table 3-1 presents a summary of the metals, organometals, and organic compounds identified as COIs for benthic invertebrates.

3.1.2 Identification of Round 2 COPCs

Round 2 COPCs were identified for each benthic invertebrate receptor by comparing maximum whole-body tissue concentrations reported (either detected or non-detected) for all COIs to aquatic tissue-residue TRVs. If the maximum tissue concentration of a receptor was greater than the aquatic tissue-residue TRV, the chemical and receptor were identified as a COPC/receptor pair for this Round 2 ERA.

The aquatic tissue-residue TRVs were developed by EPA and its partners for a screening level data evaluation in the Ecological PRE (Windward 2005; EPA 2005a). In this conservative approach, a single TRV was developed for each chemical to be protective of all aquatic species (i.e., representative of all aquatic receptors, either fish or benthic invertebrates), rather than multiple TRVs (i.e., no-observed-adverse-effect levels [NOAELs] and lowest-observed-adverse-effect levels [LOAELs]) for each

receptor group. The aquatic tissue-residue TRVs were developed using the following two-tiered approach:

- **Tier 1.** For those COIs for which a sufficient number of studies (i.e., at least 20) were included in Appendix B of the Ecological PRE (Windward 2005), the SL TRV was represented as the fifth percentile of all aquatic species LOAEL data (including fish, crayfish, and clam). This approach (i.e., using species sensitivity distribution) was intended to define a tissue-residue concentration that would be protective of 95% of aquatic species. The TRV technical memorandum in Appendix B of the Ecological PRE presents the results of the entire literature search process and a detailed discussion of how each TRV was derived from the reviewed studies.

  For those COIs for which fewer than 20 studies were identified in Appendix B of the Ecological PRE,, the fifth percentile tissue screening concentration reported in Dyer et al. (2000) was selected as the TRV. Literature-derived values included in Dyer et al. (2000) were based on single chemical laboratory tests that reported whole-body tissue concentrations associated with adverse effects on survival, reproduction, growth, behavior, and morphology. Whole-body tissue concentrations associated with effects adverse effects on biochemical or physiological endpoints were not used, nor were references that evaluated the toxicity of chemical mixtures (Dyer et al. 2000).

- **Tier 2.** For those COIs for which a sufficient amount of literature was not available either fewer than 20 studies were reviewed in Appendix B of the Ecological PRE or no fifth percentile screening was reported in Dyer et al. (2000) – the tissue residue TRV was calculated as the product of EPA AWQC and a bioconcentration factor (BCF).[4]

  The Tier 2 approach is a simple bioaccumulation model that uses an AWQC concentration multiplied by a BCF to develop TRVs. AWQC (EPA 2002) were used, when available. When current AWQC concentrations were not available, AWQC concentrations presented in the EPA Gold Book (the mid-1980s version of the national aquatic life criteria) were used, as provided by EPA were used (2005a). BCFs were based on aquatic species BCFs using K_{OW}s (octanol-water partitioning

[4] For 4-methylphenol, no AWQC was available, and the aquatic tissue-residue TRV was based on the lowest measured adverse effect tissue residue (LOAEL) presented in the Appendix B of the Ecological PRE (Windward 2005).

coefficients) developed according the methodology in Dyer et al. (2000) and Devillers (1996), as provided by EPA (2005a).

Table 3-2 presents the tissue-residue TRVs used to screen the invertebrates COIs. TRVs were available for 71 of the 78 individual chemicals identified as COIs. In addition, TRVs were developed for total polychlorinated biphenyls (PCBs) and for 2,3,7,8-tetrachlorodibenzo-*p*-dioxin as a representative for dioxins and furans. Because TRVs were not available for 2,4′-DDD, 2,4′-DDE, and 2,4′-DDT, the three pesticide isomers were assessed as part of total DDTs. No TRVs were available for butyltin, dibutyltin, and tetrabutyltin. However, because tributyltin is the most toxic chemical, the assessment of tributyltin is assumed to be protective of the other butyltins. No tissue-residue TRVs were available for 2-methylnaphthalene and benzyl alcohol.

Eighty-five COIs were detected in the field-collected clam tissue samples (Table 3-3); and of these chemicals, nine were identified as COPCs because the maximum concentration exceeded the aquatic TRV. The COPCs included three metals (cadmium, copper, and zinc), tributyltin, and five organics (total polycyclic aromatic hydrocarbons [PAHs], dibutyl phthalate total PCBs, 4,4′-DDD, and total DDTs). Beta-hexachlorocyclohexane (HCH) was identified as a Round 2 COPC based on a reporting limit (RL) exceedance of the TRV and therefore was not carried forward as a Round 2 COPC.

Fifty-two COIs were detected in the field-collected crayfish tissue samples; and of these chemicals, only copper was identified as a Round 2 COPC (maximum concentration exceeding the aquatic TRV) (Table 3-4).

Invertebrate[5] tissue samples collected with multiplate samplers were analyzed for a reduced number of chemicals because of limited tissue mass. Invertebrate tissue samples were collected at 10 locations in the Study Area. However, because of limited tissue biomass, samples from five locations were composited into two samples. The resulting seven samples were analyzed for dioxins and furans, DDTs, PCBs, and lipids. In addition, two of the tissue samples were analyzed for metals. Of these chemicals, 53 were detected (and included as COIs), but only copper was identified as a COPC (maximum concentration exceeding the aquatic TRV) (Table 3-5).

Eighty-five COIs were detected in the laboratory-exposed clam tissue samples; and of these chemicals, six were identified as Round 2 COPCs (maximum concentration exceeding the aquatic TRV). The Round 2 COPCs included copper, tributyltin, and four organics (total PAHs, 4,4′-DDD, total DDTs, and bis(2-ethylhexyl) phthalate [BEHP]) (Table 3-6).

[5] Epibenthic invertebrates and zooplankton

Eighty-six COIs were detected in the laboratory-exposed worm tissue samples; and of these chemicals, 16 were identified as Round 2 COPCs (maximum concentration exceeding the aquatic TRV). The Round 2 COPCs included 4 metals (arsenic, cadmium, copper, and zinc), tributyltin, and 11 organics (total PCBs, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, pyrene, total PAHs, 4,4′-DDD, total DDTs, and dibutyl phthalate) (Table 3-7).

A summary of the 17 Round 2 COPCs identified based on the tissue-residue approach using benthic invertebrate tissue data are presented in Table 3-8. The risks from these 17 COPCs are further evaluated in Section 3.0 of the Round 2 ERA.

3.2 PREDICTED TISSUE ASSESSMENT

The third LOE for evaluating benthic invertebrate tissue data involved comparing predicted tissue concentrations to tissue-based TRVs in order to ensure that no potential COIs were missed in the empirical tissue assessment (Section 3.1) due to the limited number of tissue samples analyzed.

The predicted tissue LOE was developed in accordance with the draft EPA guidance (Burkhard 2006). The co-located sediment and tissue data for field-collected clams, field-collected crayfish, laboratory-exposed clams, and laboratory-exposed worms were used to develop biota-sediment accumulation factors (BSAFs). Only chemicals with detected concentrations in both sediment and tissue samples were included.[6] Sediment and tissue concentrations of organic COIs were organic carbon and lipid normalized, respectively. Table 3-9 presents a summary of metals and organics identified as COIs based on the four tissue types. The co-located data were visually and statistically evaluated using scatterplots of tissue concentration versus sediment concentrations and BSAF versus sediment concentration. If the BSAF was found to be independent of sediment concentration, the average was determined of the BSAFs calculated for each co-located sediment and tissue concentration. If no relationship or an inverse relationship existed between the sediment and tissue concentration no BSAF was developed. If the BSAF varied with sediment concentration, linear and log-linear regression models were fitted to the co-located data and the best fitting model[7] was selected. This approach was performed for all COIs except, if the BSAF decreased as the sediment concentration increased and the tissue concentrations at the higher sediment concentration were non-detects, in which case a BSAF was not determined. A consistent log-linear increase in tissue concentration with increased sediment concentration was observed for PAHs and pesticides overall. BSAFs were developed for these chemicals although relationships were weak for several individual PAHs and pesticides.

[6] PCBs, dioxins and furans, and DDTs were evaluated using the FWM, see Appendix E .
[7] The best fitted model was determined based on R^2 values, visual inspection of residuals, and distribution of the independent variable.

COPCs were identified by multiplying the 95th percentile of the site-wide sediment concentration by the BSAF and compare the result to the aquatic tissue TRV. If the predicted 95th percentile tissue concentration exceeded the TRV the chemical was retained as a COPC.

Four chemicals, cadmium, zinc, TBT, and total PAHs , were identified as Round 2 COPCs based on the co-located field-collected clam tissue and sediment data (Table 3-10). Six chemicals, copper, TBT, pyrene, total PAHs, beta-HCH, and endrin, were identified as COPCs based on the co-located laboratory-exposed clam tissue and sediment data (Table 3-11). Eight chemicals, antimony, arsenic, TBT, benzo(a)pyrene, pyrene, total PAHs, beta-HCH, and endrin, were identified as Round 2 COPCs based on the co-located worm tissue and sediment data (Table 3-12). No COPC was identified based on the crayfish data (Table 3-13).

A summary of the 11 Round 2 COPCs identified based on the predicted tissue-residue approach is presented in Table 3-14. The risks from these 11 Round 2 COPCs are further evaluated in Section 3.0 of Appendix G.

4.0 WATER ASSESSMENT

Exposure and effect to the benthic community was assessed by comparing water chemistry concentrations to AWQC or TRVs available in the literature. Water data collected in Portland Harbor includes near-bottom surface water and transition zone water (TZW). These two datasets are evaluated as two separate lines of evidence. For hydrophobic organics measured in TZW, a third line of evidence was evaluated using equilibrium partitioning. The first two lines of evidence were also used to assess risks to shellfish and crayfish. This section presents the initial screening process for near-bottom surface water (Section 4.1) and TZW (Section 4.2). In addition, the equilibrium partitioning evaluation is presented in Section 4.2.3.

4.1 SURFACE WATER

This section presents the initial screening process for near-bottom surface water assessment including screening of the water data (Section 4.1.1) and COPC identification (Section 4.1.2).

4.1.1 Selection of COIs

Chemicals were identified as COIs for benthic invertebrates if they were detected in any near-bottom surface water sample excluding crustal elements (i.e., aluminum; see Section 2.0 of Appendix G). Table 4-1 presents the 98 chemicals detected in near-bottom surface water samples and identified as surface water COIs for benthic invertebrates.

4.1.2 Identification of Round 2 COPCs

Round 2 COPCs identified by comparing maximum detected COI concentrations in near-bottom surface water samples to the chronic Eco SL. If the maximum concentration was greater than the Eco SL, the chemical was identified as a Round 2 COPC for benthic invertebrates.

A review of water quality regulatory benchmarks and literature-based thresholds was conducted to develop ecological screening levels (Eco SLs) for chemicals in water. Eco SLs were developed for all benthic invertebrate surface water COIs except individual dioxins and furans other than 2,3,7,8-TCDD because no data were available. Eco SLs, presented in Attachment G3, were developed by LWG and revised based on EPA's comments (EPA 2006). LWG adopted EPA's proposed hierarchy for Eco SL selection (LWG 2006) and in accordance with EPA's comments, a UF of 50 was used to calculate a chronic screening value from an acute screening value when no chronic data were available. There is high uncertainty in the use of a UF of 50, and LWG recommends the use of a range of UFs for the BERA. The Eco SLs are considered to be protective of all aquatic receptors, including benthic invertebrates, fish, and amphibians.

Based on the results of the surface water screening, the following 10 chemicals were identified as COPCs based on maximum surface water concentrations greater than chronic Eco SLs (Table 4-2):

- Zinc (dissolved)
- Benzo(a)anthracene
- Benzo(a)pyrene
- 4-Chloro-3-methylphenol
- Total PCBs
- 2,4′-DDD
- 2,4′-DDT
- 4,4′-DDD
- 4,4′-DDT
- Total DDTs

4.2 TRANSITION ZONE WATER

This section presents the initial screening process for the TZW assessment including selection of COIs (Section 4.2.1), identification of COPCs (Section 4.2.2), and equilibrium partitioning evaluation (Section 4.2.3).

4.2.1 Selection of COIs

Chemicals were identified as COIs for benthic invertebrates if they were detected in any TZW sample collected at a depth ≤ 38 cm (in the ERA dataset), excluding crustal elements (i.e., aluminum, beryllium, cobalt, iron, magnesium, manganese and potassium; see Section 2.0 of Appendix G). Table 4-3 lists the metals, organometals, and organic compounds identified as COIs in TZW.

4.2.2 Identification of Round 2 COPCs

TZW chemicals were identified as Round 2 COPCs if the maximum concentration exceeded its respective Eco SL. Eco SLs, presented in Attachment G3, were developed by LWG and revised based on EPA's comments (EPA 2006). LWG adopted EPA's proposed hierarchy for Eco SL selection (LWG 2006). Eco SLs were available for all TZW COIs, except for calcium, titanium, diesel-range hydrocarbons, gasoline-range hydrocarbons, residual-range hydrocarbons, total petroleum hydrocarbons (TPH), and individual dioxins and furans. The Eco SLs are considered to be protective of all aquatic receptors, including benthic invertebrates, fish, and amphibians.

Fifty-three Round 2 COPCs were identified based on all TZW data collected (Table 4-4). These included 8 metals, 2 herbicides (dalapon and Silvex™), 16 PAHs, 6 pesticides (2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDE, 4,4′-DDT, and total DDTs),

3 semivolatile organic compounds (SVOCs) (1,2-dichlorobenzene, 1,4-dichlorobenzene, and dibenzofuran), 16 volatile organic compounds (VOCs), perchlorate, and cyanide.

4.2.3 Equilibrium Partitioning Evaluation

For the hydrophobic organic COIs that were not identified as TZW COPCs and for which K_{OC} values were available, an equilibrium partitioning evaluation was conducted to determine whether or not the COI was present within the Study Area at concentrations that could result in exceedances of water SLs. The COIs included in the equilibrium partitioning evaluation were one PAH (i.e., acenaphthylene) and six VOCs (i.e., 1,1-dichloroethane, 1,2-dichloroethane, acetone, chloroform, methylene chloride, and trans-1,2-dichloroethene).

For each of these COIs, the site-wide maximum organic carbon (OC)-normalized sediment concentration was identified. The OC-normalized sediment concentration was divided by the OC-normalized partition coefficient (K_{OC}) in order to calculate the maximum predicted TZW concentration. Exceedance factors were then calculated using the maximum estimated concentration and the chronic and acute water screening levels. Two Round 2 COPCs, acenaphthene and acetone, were identified from the equilibrium partitioning evaluation where the maximum predicted TZW concentration exceeded the chronic Eco SL (Table 4-5).

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TABLES

Table 3-1. Chemicals Identified as COIs for Benthic Invertebrate Receptors

Chemicals of Interest	
Metals	
Antimony	Mercury
Arsenic	Nickel
Cadmium	Selenium
Chromium	Silver
Copper	Thallium
Lead	Zinc
Butyltins	
Butyltin ion	Tetrabutyltin
Dibutyltin ion	Tributyltin ion
PAHs	
2-Methylnaphthalene	Chrysene
Acenaphthene	Dibenzo(a,h)anthracene
Acenaphthylene	Fluoranthene
Anthracene	Fluorene
Benzo(a)anthracene	Indeno(1,2,3-cd)pyrene
Benzo(a)pyrene	Naphthalene
Benzo(b)fluoranthene	Phenanthrene
Benzo(g,h,i)perylene	Pyrene
Benzo(k)fluoranthene	Total PAHs
Phthalates	
BEHP	Diethyl phthalate
Dibutyl phthalate	Dimethyl phthalate
SVOCs	
Dibenzofuran	Hexachlorobutadiene
Benzyl alcohol	n-Nitrosodiphenylamine
Hexachlorobenzene	
Phenols	
2-Methylphenol	Pentachlorophenol
4-Methylphenol	Phenol
PCBs	
Total PCBs	
Dioxins and Furans	
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin	1,2,3,6,7,8-Hexachlorodibenzofuran
2,3,7,8-Tetrachlorodibenzofuran	1,2,3,7,8,9-Hexachlorodibenzofuran
1,2,3,7,8-Pentachlorodibenzo-*p*-dioxin	2,3,4,6,7,8-Hexachlorodibenzofuran
2,3,4,7,8-Pentachlorodibenzofuran	1,2,3,4,6,7,8-Heptachlorodibenzo-*p*-dioxin
1,2,3,7,8-Pentachlorodibenzofuran	1,2,3,4,6,7,8-Heptachlorodibenzofuran
1,2,3,4,7,8-Hexachlorodibenzo-*p*-dioxin	1,2,3,4,7,8,9-Heptachlorodibenzofuran

Table 3-1. Chemicals Identified as COIs for Benthic Invertebrate Receptors

Chemicals of Interest	
1,2,3,6,7,8-Hexachlorodibenzo-*p*-dioxin	Octachlorodibenzo-*p*-dioxin
1,2,3,7,8,9-Hexachlorodibenzo-*p*-dioxin	Octachlorodibenzofuran
1,2,3,4,7,8-Hexachlorodibenzofuran	
Pesticides	
2,4′-DDD	Endosulfan sulfate
2,4′-DDE	Endrin
2,4′-DDT	Endrin aldehyde
4,4′-DDD	Endrin ketone
4,4′-DDE	Heptachlor
4,4′-DDT	Heptachlor epoxide
Total DDTs	alpha-HCH
Aldrin	beta-HCH
cis-Chlordane	gamma-HCH
trans-Chlordane	delta-HCH
Total chlordane	Methoxychlor
Dieldrin	cis-Nonachlor
alpha-Endosulfan	trans-Nonachlor
beta-Endosulfan	Oxychlordane
Total endosulfan	

BEHP – bis(2-ethylhexyl) phthalate
COI – chemical of interest
HCH – hexachlorocyclohexane
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound

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Table 3-2. Benthic Invertebrate TRVs Used for Screening COIs

Chemical	Unit (ww)	TRV	Source
Metals			
Antimony	mg/kg	0.03	estimated from AWQC and BCF
Arsenic	mg/kg	1.7	Dyer et al. (2000)
Cadmium	mg/kg	0.09	Windward (2005)
Chromium	mg/kg	2.7	estimated from AWQC and BCF
Copper	mg/kg	3.1	Dyer et al. (2000)
Lead	mg/kg	2.2	Dyer et al. (2000)
Mercury	mg/kg	0.46	Dyer et al. (2000)
Nickel	mg/kg	18.4	Dyer et al. (2000)
Selenium	mg/kg	1.1	Dyer et al. (2000)
Silver	mg/kg	0.27	Dyer et al. (2000)
Thallium	mg/kg	4.6	estimated from AWQC and BCF
Zinc	mg/kg	27	Dyer et al. (2000)
Butyltins			
Tributyltin ion[a]	µg/kg	49.9	estimated from AWQC and BCF
PAHs			
2-Methylnaphthalene	µg/kg	NA	NA
Acenaphthene	µg/kg	126,000	estimated from AWQC and BCF
Acenaphthylene	µg/kg	1,000	estimated from AWQC and BCF
Anthracene	µg/kg	1,000	estimated from AWQC and BCF
Benzo(a)anthracene	µg/kg	1,000	estimated from AWQC and BCF
Benzo(a)pyrene	µg/kg	1,000	estimated from AWQC and BCF
Benzo(b)fluoranthene	µg/kg	1,000	estimated from AWQC and BCF
Benzo(g,h,i)perylene	µg/kg	1,000	estimated from AWQC and BCF
Benzo(k)fluoranthene	µg/kg	1,000	estimated from AWQC and BCF
Chrysene	µg/kg	1,000	estimated from AWQC and BCF
Dibenzo(a,h)anthracene	µg/kg	1,000	estimated from AWQC and BCF
Fluoranthene	µg/kg	18,000	estimated from AWQC and BCF
Fluorene	µg/kg	1,000	estimated from AWQC and BCF
Indeno(1,2,3-cd)pyrene	µg/kg	1,000	estimated from AWQC and BCF
Naphthalene	µg/kg	6,500	estimated from AWQC and BCF
Phenanthrene	µg/kg	12,000	estimated from AWQC and BCF
Pyrene	µg/kg	1,000	estimated from AWQC and BCF
Total PAHs	µg/kg	1,000	benzo(a)pyrene used as surrogate
Phthalates			
BEHP	µg/kg	390	Windward (2005)
Dibutyl phthalate	µg/kg	270	estimated from AWQC and BCF
Diethyl phthalate	µg/kg	220	estimated from AWQC and BCF
Dimethyl phthalate	µg/kg	110	estimated from AWQC and BCF
SVOCs			

Table 3-2. Benthic Invertebrate TRVs Used for Screening COIs

Chemical	Unit (ww)	TRV	Source
Benzyl alcohol	µg/kg	NA	NA
Dibenzofuran	µg/kg	1,000	estimated from AWQC and BCF
Hexachlorobenzene	µg/kg	490	Dyer et al. (2000)
Hexachlorobutadiene	µg/kg	26	estimated from AWQC and BCF
n-Nitrosodiphenylamine	µg/kg	88,000	estimated from AWQC and BCF
Phenols			
2-Methylphenol	µg/kg	76,500	4-methylphenol used as surrogate
4-Methylphenol	µg/kg	76,500	Windward (2005)
Pentachlorophenol	µg/kg	3,100	estimated from AWQC and BCF
Phenol	µg/kg	6,200	estimated from AWQC and BCF
PCBs			
Total PCBs	µg/kg	720	Windward (2005)
Dioxins and Furans			
2,3,7,8-tetrachlorodibenzo-*p*-dioxin[b]	pg/kg	767	Isensee (1978)
Pesticides			
2,4′-DDD	µg/kg	NA	Evaluated under total DDTs
2,4′-DDE	µg/kg	NA	Evaluated under total DDTs
2,4′-DDT	µg/kg	NA	Evaluated under total DDTs
4,4′-DDD	µg/kg	54	estimated from AWQC and BCF
4,4′-DDE	µg/kg	1,000	Dyer et al. (2000)
4,4′-DDT	µg/kg	470	Dyer et al. (2000)
Total DDTs	µg/kg	290	Windward (2005)
Aldrin	µg/kg	810	estimated from AWQC and BCF
trans-Chlordane	µg/kg	550	total chlordane used as surrogate
cis-Chlordane	µg/kg	550	total chlordane used as surrogate
Total chlordane	µg/kg	550	Dyer et al. (2000)
Dieldrin	µg/kg	220	Dyer et al. (2000)
alpha-Endosulfan	µg/kg	73	Dyer et al. (2000)
beta-Endosulfan	µg/kg	73	Dyer et al. (2000)
Total endosulfan	µg/kg	73	Dyer et al. (2000)
Endosulfan sulfate	µg/kg	73	Dyer et al. (2000)
Endrin	µg/kg	25	Dyer et al. (2000)
Endrin aldehyde	µg/kg	25	endrin used as surrogate
Endrin ketone	µg/kg	25	endrin used as surrogate
Heptachlor	µg/kg	60	estimated from AWQC and BCF
Heptachlor epoxide	µg/kg	55	estimated from AWQC and BCF
alpha-HCH	µg/kg	4.9	estimated from AWQC and BCF
beta-HCH	µg/kg	4.9	estimated from AWQC and BCF
gamma-HCH	µg/kg	23	Dyer et al. (2000)
delta-HCH	µg/kg	4.9	estimated from AWQC and BCF
Methoxychlor	µg/kg	200	Dyer et al. (2000)

Table 3-2. Benthic Invertebrate TRVs Used for Screening COIs

Chemical	Unit (ww)	TRV	Source
cis-Nonachlor	µg/kg	550	total chlordane used as surrogate
trans-Nonachlor	µg/kg	550	total chlordane used as surrogate
Oxychlordane	µg/kg	550	total chlordane used as surrogate

[a] No toxicity data were available for butyltin ion, dibutyltin ion, or tetrabutyltin; thus, only tributyltin ion was evaluated as a COI.

[b] 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin was evaluated as a representative for all dioxins and furans.

AWQC – ambient water quality criteria
BCF – bioconcentration factor
BEHP – bis(2-ethylhexyl) phthalate
COI – chemical of interest
HCH – hexachlorocyclohexane
NA – not available or not applicable
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TRV – toxicity reference value
ww – wet weight

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Table 3-3. Results of Round 2 COPC Screen for Field-Collected Clams

COI	Unit (ww)	Maximum Tissue Concentration	TRV	Round 2 COPC?[a]
Metals				
Antimony	mg/kg	0.006 U[b]	0.03	no
Arsenic	mg/kg	1.25	1.7	no
Cadmium	mg/kg	0.218	0.09	**yes**
Chromium	mg/kg	1.05 T	2.7	no
Copper	mg/kg	13.5	3.1	**yes**
Lead	mg/kg	0.316	2.2	no
Mercury	mg/kg	0.016 J	0.46	no
Nickel	mg/kg	0.485 J	18.4	no
Selenium	mg/kg	0.21	1.1	no
Silver	mg/kg	0.101	0.27	no
Thallium	mg/kg	0.0007 J	4.6	no
Zinc	mg/kg	54	27	**yes**
Butyltins				
Tributyltin ion	µg/kg	530	49.9	**yes**
PAHs				
2-Methylnaphthalene	µg/kg	33 UT[b]	NA	NA
Acenaphthene	µg/kg	61	126,000	no
Acenaphthylene	µg/kg	33 UT[b]	1,000	no
Anthracene	µg/kg	78	1,000	no
Benzo(a)anthracene	µg/kg	670	1,000	no
Benzo(a)pyrene	µg/kg	490	1,000	no
Benzo(b)fluoranthene	µg/kg	460	1,000	no
Benzo(g,h,i)perylene	µg/kg	230	1,000	no
Benzo(k)fluoranthene	µg/kg	310	1,000	no
Chrysene	µg/kg	560	1,000	no
Dibenzo(a,h)anthracene	µg/kg	43	1,000	no
Fluoranthene	µg/kg	770	18,000	no
Fluorene	µg/kg	36	1,000	no
Indeno(1,2,3-cd)pyrene	µg/kg	170	1,000	no
Naphthalene	µg/kg	33	6,500	no
Phenanthrene	µg/kg	300	12,000	no
Pyrene	µg/kg	850	1,000	no
Total PAHs	µg/kg	4,980	1,000	**yes**
Phthalates				
BEHP	µg/kg	340 UT[b]	390	no
Dibutyl phthalate	µg/kg	1,300	270	**yes**
Diethyl phthalate	µg/kg	160 U[c]	220	no
Dimethyl phthalate	µg/kg	160 U[c]	110	no
SVOCs				

Table 3-3. Results of Round 2 COPC Screen for Field-Collected Clams

COI	Unit (ww)	Maximum Tissue Concentration	TRV	Round 2 COPC?[a]
Benzyl alcohol	µg/kg	1,300 T	NA	NA
Dibenzofuran	µg/kg	33 UT[b]	1,000	no
Hexachlorobenzene	µg/kg	12 U[b]	490	no
Hexachlorobutadiene	µg/kg	23 U[b]	26	no
n-Nitrosodiphenylamine	µg/kg	160 U[c]	88,000	no
Phenols				
2-Methylphenol	µg/kg	160 U[c]	76,500	no
4-Methylphenol	µg/kg	40 U[c]	76,500	no
Pentachlorophenol	µg/kg	160 UT[c]	3,100	no
Phenol	µg/kg	2,600 T	6,200	no
PCBs				
Total PCBs	µg/kg	2,660 J	720	**yes**
Dioxins and Furans[d]				
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin	pg/kg	1.05 U[b]	767	no
Pesticides				
2,4′-DDD	µg/kg	82 T	NA	NA
2,4′-DDE	µg/kg	12.5 JT	NA	NA
2,4′-DDT	µg/kg	39 T	NA	NA
4,4′-DDD	µg/kg	160 T	54	**yes**
4,4′-DDE	µg/kg	94.5 T	1,000	no
4,4′-DDT	µg/kg	75 T	470	no
Total DDTs	µg/kg	463 J	290	**yes**
Aldrin	µg/kg	5.07	810	no
cis-Chlordane	µg/kg	5.12	550	no
trans-Chlordane	µg/kg	5.57	550	no
Total chlordane	µg/kg	16.0 J	550	no
Dieldrin	µg/kg	2.62	220	no
alpha-Endosulfan	µg/kg	1 U[b]	73	no
beta-Endosulfan	µg/kg	2.1 UT[b]	73	no
Total endosulfan	µg/kg	2.1 U[b]	73	no
Endosulfan sulfate	µg/kg	1	73	no
Endrin	µg/kg	2.4 UT[b]	25	no
Endrin aldehyde	µg/kg	1 UJ[b]	25	no
Endrin ketone	µg/kg	1 U[b]	25	no
Heptachlor	µg/kg	1 U[b]	60	no
Heptachlor epoxide	µg/kg	8 UT[b]	55	no
alpha-HCH[c]	µg/kg	1.06 U[b]	4.9	no
beta-HCH	µg/kg	8.5 UT[b]	4.9	**yes**
gamma-HCH	µg/kg	1 U[b]	23	no
delta-HCH	µg/kg	1 U[b]	4.9	no
Methoxychlor	µg/kg	1.6 UJ[b]	200	no

Table 3-3. Results of Round 2 COPC Screen for Field-Collected Clams

COI	Unit (ww)	Maximum Tissue Concentration	TRV	Round 2 COPC?[a]
cis-Nonachlor	µg/kg	12 U[b]	550	no
trans-Nonachlor	µg/kg	4.15	550	no
Oxychlordane	µg/kg	1.13 U[b]	550	no

[a] Round 2 COPCs were identified when the maximum detected concentration in tissue was greater than the SL TRV.

[b] Maximum value is a non-detected concentration; the maximum detected concentration did not exceed the SL TRV.

[c] Maximum value is a non-detected concentration; COI was not detected in tissue and therefore was not retained as a COPC.

[d] All dioxins and furans, except 1,2,3,7,8,9-hexachlorodibenzofuran, were detected in field-collected clam tissue.

BEHP – bis(2-ethylhexyl) phthalate
COI – chemical of interest
COPC – chemical of potential concern
HCH – hexachlorocyclohexane
J – estimated concentration
NA – not available or not applicable
PAH – polycyclic aromatic hydrocarbons
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
T – value calculated or selected from multiple results
TRV – toxicity reference value
U – not detected at given concentration
ww – wet weight
Bold identifies concentration that exceeds the aquatic TRV.

Table 3-4. Results of Round 2 COPC Screen for Field-Collected Crayfish

COI	Units (ww)	Maximum Tissue Concentration	TRV	Round 2 COPC?[a]
Metals				
Antimony	mg/kg	0.02 J	0.03	no
Arsenic	mg/kg	0.5 J	1.7	no
Cadmium	mg/kg	0.036 J	0.09	no
Chromium	mg/kg	0.9	2.7	no
Copper	mg/kg	17.6	3.1	**yes**
Lead	mg/kg	1.3 JT	2.2	no
Mercury	mg/kg	0.041	0.46	no
Nickel	mg/kg	0.83	18.4	no
Selenium	mg/kg	0.3 U[b]	1.1	no
Silver	mg/kg	0.0472 J	0.27	no
Thallium	mg/kg	0.0079 J	4.6	no
Zinc	mg/kg	20.3 J	27	no
Butyltins				
Tributyltin as ion	µg/kg	NA	49.9	NA
PAHs				
2-Methylnaphthalene	µg/kg	73 UT[c]	NA	NA
Acenaphthene	µg/kg	83 UT[b]	126,000	no
Acenaphthylene	µg/kg	90 UT[b]	1,000	no
Anthracene	µg/kg	37 UT[b]	1,000	no
Benzo(a)anthracene	µg/kg	80 JT	1,000	no
Benzo(a)pyrene	µg/kg	37 UT[b]	1,000	no
Benzo(b)fluoranthene	µg/kg	63 UT[b]	1,000	no
Benzo(g,h,i)perylene	µg/kg	70 UT[b]	1,000	no
Benzo(k)fluoranthene	µg/kg	77 UT[b]	1,000	no
Chrysene	µg/kg	87 T	1,000	no
Dibenzo(a,h)anthracene	µg/kg	37 UT[b]	1,000	no
Fluoranthene	µg/kg	130 T	18,000	no
Fluorene	µg/kg	70 UT[b]	1,000	no
Indeno(1,2,3-cd)pyrene	µg/kg	37 UT[b]	1,000	no
Naphthalene	µg/kg	82 UT[b]	6,500	no
Phenanthrene	µg/kg	97 T	12,000	no
Pyrene	µg/kg	83 T	1,000	no
Total PAHs	µg/kg	731 J	1,000	no
Phthalates				
BEHP	µg/kg	270 UJT[b]	390	no
Dibutyl phthalate	µg/kg	650 U[b]	270	no
Diethyl phthalate	µg/kg	950 U[b]	220	no
Dimethyl phthalate	µg/kg	190 U[b]	110	no

Table 3-4. Results of Round 2 COPC Screen for Field-Collected Crayfish

COI	Units (ww)	Maximum Tissue Concentration	TRV	Round 2 COPC?[a]
SVOCs				
Benzyl alcohol	µg/kg	370 U[b]	NA	NA
Dibenzofuran	µg/kg	66 UT[b]	1,000	no
Hexachlorobenzene	µg/kg	2 UT[b]	490	no
Hexachlorobutadiene	µg/kg	2 UJT[b]	26	no
n-Nitrosodiphenylamine	µg/kg	190 U[b]	88,000	no
Phenols				
2-Methylphenol	µg/kg	190 U[b]	76,500	no
4-methylphenol	µg/kg	190 T	76,500	no
Pentachlorophenol	µg/kg	190 UT[c]	3,100	no
Phenol	µg/kg	520	6,200	no
PCBs				
Total PCBs	µg/kg	335	720	no
Dioxins and Furans[d]				
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin	pg/kg	0.252	767	no
Pesticides				
2,4′-DDD	µg/kg	4.3 NJ	NA	NA
2,4′-DDE	µg/kg	3.3 NJ	NA	NA
2,4′-DDT	µg/kg	9.5 NJ	NA	NA
4,4′-DDD	µg/kg	17 NJ	54	no
4,4′-DDE	µg/kg	51	1,000	no
4,4′-DDT	µg/kg	14 J	470	no
Total DDTs	µg/kg	85.4 NJ	290	no
Aldrin	µg/kg	2 U[b]	810	no
cis-Chlordane	µg/kg	2 UJ[b]	550	no
trans-Chlordane	µg/kg	2.7 NJ	550	no
Total chlordane	µg/kg	4.7 NJ	550	no
Dieldrin	µg/kg	2 UJ[b]	220	no
alpha-Endosulfan	µg/kg	2 U[b]	73	no
beta-Endosulfan	µg/kg	3.1 J	73	no
Total endosulfan	µg/kg	4.1 J	73	no
Endosulfan sulfate	µg/kg	2 UJ[b]	73	no
Endrin	µg/kg	2.8 NJ	25	no
Endrin aldehyde	µg/kg	2 UJ[b]	25	no
Endrin ketone	µg/kg	2 UJ[b]	25	no
Heptachlor	µg/kg	2 U[b]	60	no
Heptachlor epoxide	µg/kg	2 U[b]	55	no
alpha-HCH	µg/kg	2 U[b]	4.9	no
beta-HCH	µg/kg	2 UJ[b]	4.9	no
gamma-HCH	µg/kg	2 U[b]	23	no
delta-HCH	µg/kg	2.8 U[b]	4.9	no

Table 3-4. Results of Round 2 COPC Screen for Field-Collected Crayfish

COI	Units (ww)	Maximum Tissue Concentration	TRV	Round 2 COPC?[a]
Methoxychlor	µg/kg	2 U[b]	200	no
cis-Nonachlor	µg/kg	2.3 U[b]	550	no
trans-Nonachlor	µg/kg	2 UJ[b]	550	no
Oxychlordane	µg/kg	2 U[b]	550	no

[a] Round 2 COPCs were identified when the maximum detected concentration in tissue was greater than the SL TRV.

[b] Maximum value is a non-detected concentration; COI was not detected in tissue and therefore was not retained as a COPC.

[c] Maximum value is a non-detected concentration; the maximum detected concentration did not exceed the SL TRV.

[d] All dioxins and furans were detected in field-collected clam tissue.

BEHP – bis(2-ethylhexyl) phthalate
COI – chemical of interest
COPC – chemical of potential concern
HCH – hexachlorocyclohexane
J – estimated concentration
N – presumptive evidence of a compound
NA – not a available or not applicable
PAH – polycyclic aromatic hydrocarbons
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
T – value calculated or selected from multiple results
TRV – toxicity reference value
U – not detected at given concentration
ww – wet weight
Bold identifies concentration that exceeds the aquatic TRV.

Table 3-5. Results of Round 2 COPC Screen for Invertebrates Collected with Multiplate Samplers

COI	Unit (ww)	Maximum Tissue Concentration	TRV	Round 2 COPC?[a]
Metals				
Antimony	mg/kg	0.0022 J	0.03	no
Arsenic	mg/kg	0.45	1.7	no
Cadmium	mg/kg	0.0366	0.09	no
Chromium	mg/kg	1.73	2.7	no
Copper	mg/kg	6 J	3.1	**yes**
Lead	mg/kg	1.06 J	2.2	no
Mercury	mg/kg	NA	0.46	NA
Nickel	mg/kg	1.12	18.4	no
Selenium	mg/kg	0.06	1.1	no
Silver	mg/kg	0.0287 J	0.27	no
Thallium	mg/kg	NA	4.6	NA
Zinc	mg/kg	24.8 J	27	no
SVOCs				
Hexachlorobenzene	µg/kg	0.544	490	no
PCBs				
Total PCBs	µg/kg	498	720	no
Dioxins and Furans[b]				
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin	pg/kg	0.329 U[c]	767	no
Pesticides				
2,4′-DDD	µg/kg	14.1	NA	NA
2,4′-DDE	µg/kg	1.28	NA	NA
2,4′-DDT	µg/kg	7.54	NA	NA
4,4′-DDD	µg/kg	30.3	54	no
4,4′-DDE	µg/kg	29.4	1,000	no
4,4′-DDT	µg/kg	12.2	470	no
Total DDTs	µg/kg	94.8	290	no
Aldrin	µg/kg	0.0872	810	no
cis-Chlordane	µg/kg	0.6	550	no
trans-Chlordane	µg/kg	0.466	550	no
Total chlordane	µg/kg	2.06 J	550	no
Dieldrin	µg/kg	0.396	220	no
alpha-Endosulfan	µg/kg	0.137 U[c]	73	no
beta-Endosulfan	µg/kg	0.156 U[c]	73	no
Total endosulfan	µg/kg	0.491 J	73	no
Endosulfan sulfate	µg/kg	0.296	73	no
Endrin	µg/kg	0.146 U[c]	25	no
Heptachlor	µg/kg	0.012 J	60	no
Heptachlor epoxide	µg/kg	0.025 J	55	no

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Table 3-5. Results of Round 2 COPC Screen for Invertebrates Collected with Multiplate Samplers

COI	Unit (ww)	Maximum Tissue Concentration	TRV	Round 2 COPC?[a]
alpha-HCH	µg/kg	0.0195 J	4.9	no
beta-HCH	µg/kg	0.0268 U[c]	4.9	no
gamma-HCH	µg/kg	0.0214 J	23	no
cis-Nonachlor	µg/kg	0.24	550	no
trans-Nonachlor	µg/kg	0.697	550	no
Oxychlordane	µg/kg	0.183	550	no

[a] Round 2 COPCs were identified when the maximum detected concentration in tissue was greater than the SL TRV.

[b] All dioxins and furans, except 1,2,3,7,8,9-hexachlorodibenzofuran and 2,3,4,6,7,8-hexachorodibenzofuran, were detected in field-collected clam tissue.

[c] Maximum value is a non-detected concentration; the maximum detected concentration did not exceed the SL TRV.

COI – chemical of interest
COPC – chemical of potential concern
HCH – hexachlorocyclohexane
J – estimated concentration
NA – not available or not applicable
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TRV – toxicity reference value
U – not detected at given concentration
ww – wet weight
Bold identifies concentration that exceeds the aquatic TRV.

Table 3-6. Results of Round 2 COPC Screen for Laboratory-Exposed Clams

COI	Unit (ww)	Maximum Tissue Concentration	TRV	Round 2 COPC?[a]
Metals				
Antimony	mg/kg	0.0035 U[b]	0.03	no
Arsenic	mg/kg	0.548	1.7	no
Cadmium	mg/kg	0.0666	0.09	no
Chromium	mg/kg	0.49	2.7	no
Copper	mg/kg	5.94 J	3.1	**yes**
Lead	mg/kg	0.221	2.2	no
Mercury	mg/kg	0.0162	0.46	no
Nickel	mg/kg	0.313 J	18.4	no
Selenium	mg/kg	0.247	1.1	no
Silver	mg/kg	0.0181	0.27	no
Thallium	mg/kg	NA	4.6	NA
Zinc	mg/kg	16.8	27	no
Butyltins				
Tributyltin ion	µg/kg	680	49.9	**yes**
PAHs				
2-Methylnaphthalene	µg/kg	1.8	NA	NA
Acenaphthene	µg/kg	13	126,000	no
Acenaphthylene	µg/kg	1.1	1,000	no
Anthracene	µg/kg	29	1,000	no
Benzo(a)anthracene	µg/kg	67	1,000	no
Benzo(a)pyrene	µg/kg	33	1,000	no
Benzo(b)fluoranthene	µg/kg	33	1,000	no
Benzo(g,h,i)perylene	µg/kg	18	1,000	no
Benzo(k)fluoranthene	µg/kg	23	1,000	no
Chrysene	µg/kg	100	1,000	no
Dibenzo(a,h)anthracene	µg/kg	1.4	1,000	no
Fluoranthene	µg/kg	390	18,000	no
Fluorene	µg/kg	11	1,000	no
Indeno(1,2,3-cd)pyrene	µg/kg	12	1,000	no
Naphthalene	µg/kg	5.1	6,500	no
Phenanthrene	µg/kg	120	12,000	no
Pyrene	µg/kg	490	1,000	no
Total PAHs	µg/kg	1320	1,000	**yes**
Phthalates				
BEHP	µg/kg	8,600	390	**yes**
Dibutyl phthalate	µg/kg	210 U[c]	270	no
Diethyl phthalate	µg/kg	19 J	220	no
Dimethyl phthalate	µg/kg	5.6 J	110	no

Table 3-6. Results of Round 2 COPC Screen for Laboratory-Exposed Clams

COI	Unit (ww)	Maximum Tissue Concentration	TRV	Round 2 COPC?[a]
SVOCs				
Benzyl alcohol	µg/kg	25 J	NA	NA
Dibenzofuran	µg/kg	1.2	1,000	no
Hexachlorobenzene	µg/kg	0.275	490	no
Hexachlorobutadiene	µg/kg	8.5 U[b]	26	no
n-Nitrosodiphenylamine	µg/kg	9.5 U[c]	88,000	no
Phenols				
2-Methylphenol	µg/kg	99	76,500	no
4-Methylphenol	µg/kg	15 U[c]	76,500	no
Pentachlorophenol	µg/kg	31 U[c]	3,100	no
Phenol	µg/kg	180	6,200	no
PCBs				
Total PCBs	µg/kg	189	720	no
Dioxins and Furans[d]				
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin	pg/kg	15.4	767	no
Pesticides				
2,4′-DDD	µg/kg	239	NA	NA
2,4′-DDE	µg/kg	2.93	NA	NA
2,4′-DDT	µg/kg	0.729	NA	NA
4,4′-DDD	µg/kg	702	54	**yes**
4,4′-DDE	µg/kg	8.55	1,000	no
4,4′-DDT	µg/kg	87.1	470	no
Total DDTs	µg/kg	1040	290	**yes**
Aldrin	µg/kg	2.14	810	no
cis-Chlordane	µg/kg	2.29	550	no
trans-Chlordane	µg/kg	2.81	550	no
Total chlordane	µg/kg	7.70 J	550	no
Dieldrin	µg/kg	4.14	220	no
alpha-Endosulfan	µg/kg	0.049 J	73	no
beta-Endosulfan	µg/kg	0.119 U[b]	73	no
Total endosulfan	µg/kg	0.149 J	73	no
Endosulfan sulfate	µg/kg	0.086 U[b]	73	no
Endrin	µg/kg	0.0616 J	25	no
Endrin aldehyde	µg/kg	0.0403 U[c]	25	no
Endrin ketone	µg/kg	0.0839 U[b]	25	no
Heptachlor	µg/kg	0.0607 U[b]	60	no
Heptachlor epoxide	µg/kg	0.094 J	55	no
alpha-HCH	µg/kg	0.0633 U[b]	4.9	no
beta-HCH	µg/kg	0.0517 U[b]	4.9	no
gamma-HCH	µg/kg	0.109 J	23	no
delta-HCH	µg/kg	0.0361 U[b]	4.9	no

Table 3-6. Results of Round 2 COPC Screen for Laboratory-Exposed Clams

COI	Unit (ww)	Maximum Tissue Concentration	TRV	Round 2 COPC?[a]
Methoxychlor	µg/kg	0.0284 U[c]	200	no
cis-Nonachlor	µg/kg	0.985	550	no
trans-Nonachlor	µg/kg	3	550	no
Oxychlordane	µg/kg	0.157 J	550	no

[a] Round 2 COPCs were identified when the maximum detected concentration in tissue was greater than the SL TRV.

[b] Maximum value is a non-detected concentration; the maximum detected concentration did not exceed the SL TRV.

[c] Maximum value is a non-detected concentration; COI was not detected in tissue and therefore was not retained as a COPC.

[d] All dioxins and furans were detected in field-collected clam tissue.

BEHP – bis(2-ethylhexyl) phthalate
COI – chemical of interest
COPC – chemical of potential concern
HCH – hexachlorocyclohexane
J – estimated concentration
NA – not available or not applicable
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
U – not detected at given concentration
ww – wet weight
Bold identifies concentration that exceeds the aquatic TRV.

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Table 3-7. Results of Round 2 COPC Screen for Laboratory-Exposed Worms

COI	Unit (ww)	Maximum Tissue Concentration	TRV	Round 2 COPC?[a]
Metals				
Antimony	mg/kg	0.014 J	0.03	no
Arsenic	mg/kg	3.04	1.7	**yes**
Cadmium	mg/kg	0.254	0.09	**yes**
Chromium	mg/kg	0.89	2.7	no
Copper	mg/kg	20.2	3.1	**yes**
Lead	mg/kg	0.847 T	2.2	no
Mercury	mg/kg	0.0105	0.46	no
Nickel	mg/kg	0.535	18.4	no
Selenium	mg/kg	0.37	1.1	no
Silver	mg/kg	0.0133	0.27	no
Thallium	mg/kg	NA	4.6	NC
Zinc	mg/kg	31.5	27	**yes**
Butyltins				
Tributyltin ion	µg/kg	1,700	49.9	**yes**
PAHs				
2-Methylnaphthalene	µg/kg	43	NA	NA
Acenaphthene	µg/kg	520	126,000	no
Acenaphthylene	µg/kg	29	1,000	no
Anthracene	µg/kg	650	1,000	no
Benzo(a)anthracene	µg/kg	2,600	1,000	**yes**
Benzo(a)pyrene	µg/kg	1,500	1,000	**yes**
Benzo(b)fluoranthene	µg/kg	1,600	1,000	**yes**
Benzo(g,h,i)perylene	µg/kg	720	1,000	no
Benzo(k)fluoranthene	µg/kg	1,500	1,000	**yes**
Chrysene	µg/kg	3,900	1,000	**yes**
Dibenzo(a,h)anthracene	µg/kg	95	1,000	no
Fluoranthene	µg/kg	10,000	18,000	no
Fluorene	µg/kg	300	1,000	no
Indeno(1,2,3-cd)pyrene	µg/kg	670	1,000	no
Naphthalene	µg/kg	31	6,500	no
Phenanthrene	µg/kg	2,500	12,000	no
Pyrene	µg/kg	11,000	1,000	**yes**
Total PAHs	µg/kg	37,300	1,000	**yes**
Phthalates				
BEHP	µg/kg	220 J	390	no
Dibutyl phthalate	µg/kg	470 J	270	**yes**
Diethyl phthalate	µg/kg	28 U[b]	220	no
Dimethyl phthalate	µg/kg	11 U[b]	110	no

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Table 3-7. Results of Round 2 COPC Screen for Laboratory-Exposed Worms

COI	Unit (ww)	Maximum Tissue Concentration	TRV	Round 2 COPC?[a]
SVOCs				
Benzyl alcohol	µg/kg	4,900 J	NA	NA
Dibenzofuran	µg/kg	27	1,000	no
Hexachlorobenzene	µg/kg	2.69	490	no
Hexachlorobutadiene	µg/kg	17 U[c]	26	no
n-Nitrosodiphenylamine	µg/kg	190	88,000	no
Phenols				
2-Methylphenol	µg/kg	350	76,500	no
4-Methylphenol	µg/kg	290	76,500	no
Pentachlorophenol	µg/kg	31 U[b]	3,100	no
Phenol	µg/kg	50 J	6,200	no
PCBs				
Total PCBs	µg/kg	4,310 J	720	**yes**
Dioxins and Furans[d]				
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin	pg/kg	163	767	no
Pesticides				
2,4′-DDD	µg/kg	204	NA	NA
2,4′-DDE	µg/kg	52.7	NA	NA
2,4′-DDT	µg/kg	4.27 U[c]	NA	NA
4,4′-DDD	µg/kg	1,060	54	**yes**
4,4′-DDE	µg/kg	171	1,000	no
4,4′-DDT	µg/kg	37.9	470	no
Total DDTs	µg/kg	1,490	290	**yes**
Aldrin	µg/kg	37	810	no
cis-Chlordane	µg/kg	28.6	550	no
trans-Chlordane	µg/kg	30.5	550	no
Total chlordane	µg/kg	71.9	550	no
Dieldrin	µg/kg	26.7	220	no
alpha-Endosulfan	µg/kg	0.181 J	73	no
beta-Endosulfan	µg/kg	0.492	73	no
Total endosulfan	µg/kg	1.42 J	73	no
Endosulfan sulfate	µg/kg	0.743	73	no
Endrin	µg/kg	0.157 J	25	no
Endrin aldehyde	µg/kg	0.0384 U[b]	25	no
Endrin ketone	µg/kg	0.08 U[c]	25	no
Heptachlor	µg/kg	0.0224 U[c]	60	no
Heptachlor epoxide	µg/kg	0.0711 J	55	no
alpha-HCH	µg/kg	0.0633 U[c]	4.9	no
beta-HCH	µg/kg	0.31 U[c]	4.9	no
delta-HCH	µg/kg	0.0195 U[c]	4.9	no
gamma-HCH	µg/kg	0.215 U[c]	23	no

Table 3-7. Results of Round 2 COPC Screen for Laboratory-Exposed Worms

COI	Unit (ww)	Maximum Tissue Concentration	TRV	Round 2 COPC?[a]
Methoxychlor	µg/kg	0.486 U[b]	200	no
cis-Nonachlor	µg/kg	3.0	550	no
trans-Nonachlor	µg/kg	9.8	550	no
Oxychlordane	µg/kg	0.334 J	550	no

[a] Round 2 COPCs were identified when the maximum detected concentration in tissue was greater than the SL TRV.

[b] Maximum value is a non-detected concentration; COI was not detected in tissue and therefore was not retained as a COPC.

[c] Maximum value is a non-detected concentration; the maximum detected concentration did not exceed the SL TRV.

[d] All dioxins and furans were detected in field-collected clam tissue.

BEHP – bis(2-ethylhexyl) phthalate
COI – chemical of interest
COPC – chemical of potential concern
HCH – hexachlorocyclohexane
J – estimated concentration
NA – not available or not applicable
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
T – value calculated or selected from multiple results
TRV – toxicity reference value
U – not detected at given concentration
ww – wet weight
Bold identifies concentration that exceeds the aquatic TRV.

Table 3-8. Summary of Round 2 COPCs for Benthic Invertebrates

Round 2 COPC	Field-Collected Benthic Invertebrates			Laboratory-Exposed Benthic Invertebrates	
	Clam	Crayfish	Multiplate Invertebrate[a]	Clam	Worm
Metals					
Arsenic					X
Cadmium	X				X
Copper	X	X	X	X	X
Zinc	X				X
Butyltins					
Tributyltin ion	X			X	X
PAHs					
Benzo(a)anthracene					X
Benzo(a)pyrene					X
Benzo(b)fluoranthene					X
Benzo(k)fluoranthene					X
Chrysene					X
Pyrene					X
Total PAHs	X			X	X
Phthalates					
BEHP				X	
Dibutyl phthalate	X				X
PCBs					
Total PCBs	X				X
Pesticides					
4,4′-DDD	X			X	X
Total DDTs	X			X	X

[a] Epibenthic invertebrates and zooplankton.
BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl

Table 3-9. Chemicals Identified as COIs Based on the Predicted Tissue Approach

Chemicals of Interest	
Metals	
Antimony	Mercury
Arsenic	Nickel
Cadmium	Silver
Chromium	Thallium
Copper	Zinc
Lead	
Butyltins	
Tributyltin ion	
PAHs	
2-Methylnaphthalene	Chrysene
Acenaphthene	Dibenzo(a,h)anthracene
Acenaphthylene	Fluoranthene
Anthracene	Fluorene
Benzo(a)anthracene	Indeno(1,2,3-cd)pyrene
Benzo(a)pyrene	Naphthalene
Benzo(b)fluoranthene	Phenanthrene
Benzo(g,h,i)perylene	Pyrene
Benzo(k)fluoranthene	Total PAHs
Phthalates	
BEHP	Diethyl phthalate
Dibutyl phthalate	Dimethyl phthalate
SVOCs	
Dibenzofuran	Hexachlorobutadiene
Hexachlorobenzene	n-Nitrosodiphenylamine
Phenols	
4-Methylphenol	Phenol
Pentachlorophenol	
Pesticides	
Aldrin	Heptachlor
cis-Chlordane	Heptachlor epoxide
trans-Chlordane	alpha-HCH
Total chlordane	beta-HCH
Dieldrin	gamma-HCH
alpha-Endosulfan	delta-HCH
beta-Endosulfan	Methoxychlor
Total endosulfan	cis-Nonachlor
Endosulfan sulfate	trans-Nonachlor
Endrin	Oxychlordane
Endrin ketone	

COI – chemicals of interest
HCH –hexachlorocyclohexane
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound

Table 3-10. Results of Round 2 COPC Predicted Tissue Screen for Field-Collected Clams

COI	Unit	Sediment-Tissue Relationship?	Maximum Predicted Tissue Concentration[a]	TRV	Round 2 COPC?[b]
Metals					
Antimony	mg/kg ww	no	NA	0.03	NA
Arsenic	mg/kg ww	no	NA	1.7	NA
Cadmium	mg/kg ww	yes	1.07	0.09	**yes**
Chromium	mg/kg ww	no	NA	2.7	NA
Copper	mg/kg ww	no	NA	3.1	NA
Lead	mg/kg ww	yes	0.143	2.2	no
Mercury	mg/kg ww	yes	0.0108	0.46	no
Nickel	mg/kg ww	no	NA	18.4	NA
Selenium	mg/kg ww	no	NA	1.1	NA
Silver	mg/kg ww	no	NA	0.27	NA
Thallium	mg/kg ww	no	NA	4.6	NA
Zinc	mg/kg ww	yes	40.3	27	**yes**
Butyltins					
Tributyltin ion	µg/kg lipid	yes	3.61	2.19	**yes**
PAHs					
2-Methylnaphthalene	µg/kg lipid	no	NA	NA	NA
Acenaphthene	µg/kg lipid	yes	0.661	5530	no
Acenaphthylene	µg/kg lipid	yes	0.3534	43.9	no
Anthracene	µg/kg lipid	yes	3.58	43.9	no
Benzo(a)anthracene	µg/kg lipid	yes	19.1	43.9	no
Benzo(a)pyrene	µg/kg lipid	yes	6.43	43.9	no
Benzo(b)fluoranthene	µg/kg lipid	yes	9.25	43.9	no
Benzo(g,h,i)perylene	µg/kg lipid	yes	3.23	43.9	no
Benzo(k)fluoranthene	µg/kg lipid	yes	3.70	43.9	no
Chrysene	µg/kg lipid	yes	15.0	43.9	no
Dibenzo(a,h)anthracene	µg/kg lipid	yes	0.737	43.9	no
Fluoranthene	µg/kg lipid	yes	29.4	791	no
Fluorene	µg/kg lipid	yes	0.593	43.9	no
Indeno(1,2,3-cd)pyrene	µg/kg lipid	yes	1.22	43.9	no
Naphthalene	µg/kg lipid	yes	0.628	2860	no
Phenanthrene	µg/kg lipid	yes	9.56	527	no
Pyrene	µg/kg lipid	yes	37.4	43.9	no
Total PAHs	µg/kg lipid	yes	155	43.9	**yes**
Phthalates					
BEHP	µg/kg lipid	yes	8.60	17.1	no
Dibutyl phthalate	µg/kg lipid	no	NA	11.9	NA
Diethyl phthalate	µg/kg lipid	no	NA	9.66	NA
Dimethyl phthalate	µg/kg lipid	no	NA	4.83	NA

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Table 3-10. Results of Round 2 COPC Predicted Tissue Screen for Field-Collected Clams

COI	Unit	Sediment-Tissue Relationship?	Maximum Predicted Tissue Concentration[a]	TRV	Round 2 COPC?[b]
SVOCs					
Dibenzofuran	µg/kg lipid	no	NA	43.9	NA
Hexachlorobenzene	µg/kg lipid	no	NA	21.5	NA
Hexachlorobutadiene	µg/kg lipid	no	NA	1.14	NA
Hexachloroethane	µg/kg lipid	no	NA	2060	NA
n-Nitrosodiphenylamine	µg/kg lipid	no	NA	3870	NA
Phenols					
2-Methylphenol	µg/kg lipid	no	NA	3360	NA
4-Methylphenol	µg/kg lipid	no	NA	3360	NA
Pentachlorophenol	µg/kg lipid	no	NA	136	NA
Phenol	µg/kg lipid	no	NA	272	NA
Pesticides					
Aldrin	µg/kg lipid	yes	0.0905	35.6	no
cis-Chlordane	µg/kg lipid	yes	0.0962	24.2	no
trans-Chlordane	µg/kg lipid	yes	0.102	24.2	NA
Total chlordane	µg/kg lipid	yes	0.344	24.2	no
Dieldrin	µg/kg lipid	no	NA	9.66	NA
alpha-Endosulfan	µg/kg lipid	no	NA	3.21	NA
beta-Endosulfan	µg/kg lipid	no	NA	3.21	NA
Total endosulfan	µg/kg lipid	no	NA	3.21	NA
Endosulfan sulfate	µg/kg lipid	no	NA	3.21	NA
Endrin	µg/kg lipid	yes	0.0495	1.10	no
Endrin aldehyde	µg/kg lipid	no	NA	1.10	NA
Endrin ketone	µg/kg lipid	yes	0.134	1.10	no
Heptachlor	µg/kg lipid	yes	0.557	2.64	no
Heptachlor epoxide	µg/kg lipid	no	NA	2.42	NA
alpha-Hexachlorocyclohexane	µg/kg lipid	no	NA	0.215	NA
beta-Hexachlorocyclohexane	µg/kg lipid	no	NA	0.215	NA
delta-Hexachlorocyclohexane	µg/kg lipid	no	NA	0.215	NA
gamma-Hexachlorocyclohexane	µg/kg lipid	no	NA	1.01	NA
Methoxychlor	µg/kg lipid	no	NA	8.79	NA
cis-Nonachlor	µg/kg lipid	yes	0.0293	24.2	no
trans-Nonachlor	µg/kg lipid	yes	0.0911	24.2	no
Oxychlordane	µg/kg lipid	yes	8.13	24.2	no

[a] Predicted tissue concentration is based on the site-specific BSAF and the 95th percentile of detected site-wide sediment concentrations.

[b] Round 2 COPCs were identified when the predicted tissue concentration was greater than the SL TRV.

COI – chemicals of interest
COPC –chemical of potential concern
HCH –hexachlorocyclohexane
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound

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Table 3-11. Results of Round 2 COPC Predicted Tissue Screen for Laboratory-Exposed Clams

COI	Unit	Sediment-Tissue Relationship?	Maximum Predicted Tissue Concentration[a]	TRV	Round 2 COPC?[b]
Metals					
Antimony	mg/kg ww	no	NA	0.03	NA
Arsenic	mg/kg ww	no	NA	1.7	NA
Cadmium	mg/kg ww	yes	0.06	0.09	no
Chromium	mg/kg ww	yes	0.26	2.7	no
Copper	mg/kg ww	yes	4.65	3.1	**yes**
Lead	mg/kg ww	yes	0.13	2.2	no
Mercury	mg/kg ww	no	NA	0.46	NA
Nickel	mg/kg ww	no	NA	18.4	NA
Selenium	mg/kg ww	no	NA	1.1	NA
Silver	mg/kg ww	no	NA	0.27	NA
Thallium	mg/kg ww	NA	NA	4.6	NA
Zinc	mg/kg ww	no	NA	27	NA
Butyltins					
Tributyltin ion	µg/kg lipid	yes	17.09	4.97	**yes**
PAHs					
2-Methylnaphthalene	µg/kg lipid	Yes	0.18	NA	NA
Acenaphthene	µg/kg lipid	yes	17.3	12500	no
Acenaphthylene	µg/kg lipid	yes	3.91	99.5	no
Anthracene	µg/kg lipid	yes	23.1	99.5	no
Benzo(a)anthracene	µg/kg lipid	yes	19.5	99.5	no
Benzo(a)pyrene	µg/kg lipid	yes	10.2	99.5	no
Benzo(b)fluoranthene	µg/kg lipid	yes	14.8	99.5	no
Benzo(g,h,i)perylene	µg/kg lipid	yes	7.91	99.5	no
Benzo(k)fluoranthene	µg/kg lipid	yes	4.91	99.5	no
Chrysene	µg/kg lipid	yes	40.4	99.5	no
Dibenzo(a,h)anthracene	µg/kg lipid	no	NA	99.5	no
Fluoranthene	µg/kg lipid	yes	192	1790	no
Fluorene	µg/kg lipid	yes	1.10	99.5	no
Indeno(1,2,3-cd)pyrene	µg/kg lipid	yes	0.57	99.5	no
Naphthalene	µg/kg lipid	yes	1.76	647	no
Phenanthrene	µg/kg lipid	yes	10.9	1190	no
Pyrene	µg/kg lipid	yes	189	99.5	**yes**
Total PAHs	µg/kg lipid	yes	607	99.5	**yes**
Phthalates					
BEHP	µg/kg lipid	no	NA	38.8	NA
Dibutyl phthalate	µg/kg lipid	no	NA	26.9	NA
Diethyl phthalate	µg/kg lipid	no	NA	21.9	NA
Dimethyl phthalate	µg/kg lipid	no	NA	10.9	NA

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Table 3-11. Results of Round 2 COPC Predicted Tissue Screen for Laboratory-Exposed Clams

COI	Unit	Sediment-Tissue Relationship?	Maximum Predicted Tissue Concentration[a]	TRV	Round 2 COPC?[b]
SVOCs					
Dibenzofuran	µg/kg lipid	yes	0.45	99.5	no
Hexachlorobenzene	µg/kg lipid	yes	0.07	48.8	no
Hexachlorobutadiene	µg/kg lipid	yes	0.34	2.59	no
Hexachloroethane	µg/kg lipid	no	NA	4677	NA
n-Nitrosodiphenylamine	µg/kg lipid	no	NA	876	NA
Phenols					
2-Methylphenol	µg/kg lipid	no	NA	7610	NA
4-Methylphenol	µg/kg lipid	no	NA	7610	NA
Pentachlorophenol	µg/kg lipid	no	NA	308	NA
Phenol	µg/kg lipid	no	NA	617	NA
Pesticides					
Aldrin	µg/kg lipid	yes	0.50	80.6	no
cis-Chlordane	µg/kg lipid	yes	0.48	54.7	no
trans-Chlordane	µg/kg lipid	yes	0.51	54.7	no
Total chlordane	µg/kg lipid	yes	1.80	54.7	no
Dieldrin	µg/kg lipid	yes	0.22	21.9	no
alpha-Endosulfan	µg/kg lipid	yes	0.72	7.26	no
beta-Endosulfan	µg/kg lipid	yes	0.36	7.26	no
Total endosulfan	µg/kg lipid	no	NA	7.26	NA
Endosulfan sulfate	µg/kg lipid	no	NA	7.26	NA
Endrin	µg/kg lipid	yes	7.57	2.49	**yes**
Endrin aldehyde	µg/kg lipid	no	NA	2.49	NA
Endrin ketone	µg/kg lipid	yes	0.76	2.49	no
Heptachlor	µg/kg lipid	yes	1.48	5.97	no
Heptachlor epoxide	µg/kg lipid	no	NA	5.47	NA
alpha-Hexachlorocyclohexane	µg/kg lipid	no	NA	0.486	NA
beta-Hexachlorocyclohexane	µg/kg lipid	yes	1.39	0.488	**yes**
gamma-Hexachlorocyclohexane	µg/kg lipid	no	NA	2.29	NA
delta-Hexachlorocyclohexane	µg/kg lipid	no	NA	0.486	NA
Methoxychlor	µg/kg lipid	no	NA	19.9	NA
cis-Nonachlor	µg/kg lipid	yes	0.46	54.7	no
trans-Nonachlor	µg/kg lipid	yes	0.51	54.7	no
Oxychlordane	µg/kg lipid	yes	15.71	54.7	no

[a] Predicted tissue concentration is based on the site-specific BSAF and the 95th percentile of detected site-wide sediment concentrations.

[b] Round 2 COPCs were identified when the predicted tissue concentration was greater than the SL TRV.

COI – chemicals of interest
COPC –chemical of potential concern
HCH –hexachlorocyclohexane
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound

Table 3-12. Results of Round 2 COPC Predicted Tissue Screen for Laboratory-Exposed Worms

COI	Unit	Sediment-Tissue Relationship?	Maximum Predicted Tissue Concentration[a]	TRV	Round 2 COPC?[b]
Metals					
Antimony	mg/kg ww	yes	0.0881	0.03	**yes**
Arsenic	mg/kg ww	yes	2.36	1.7	**yes**
Cadmium	mg/kg ww	no	NA	0.09	NA
Chromium	mg/kg ww	no	NA	2.7	NA
Copper	mg/kg ww	no	NA	3.1	NA
Lead	mg/kg ww	yes	0.634	2.2	no
Mercury	mg/kg ww	yes	0.00743	0.46	no
Nickel	mg/kg ww	no	NA	18.4	NA
Selenium	mg/kg ww	no	NA	1.1	NA
Silver	mg/kg ww	no	NA	0.27	NA
Thallium	mg/kg ww	NA	NA	4.6	NA
Zinc	mg/kg ww	no	NA	27	NA
Butyltins					
Tributyltin ion	µg/kg lipid	yes	14.01	2.10	**yes**
PAHs					
2-Methylnaphthalene	µg/kg lipid	yes	0.479	NA	NA
Acenaphthene	µg/kg lipid	yes	32.6	12500	no
Acenaphthylene	µg/kg lipid	yes	0.736	99.5	no
Anthracene	µg/kg lipid	yes	10.6	99.5	no
Benzo(a)anthracene	µg/kg lipid	yes	23.6	99.5	no
Benzo(a)pyrene	µg/kg lipid	yes	248	99.5	**yes**
Benzo(b)fluoranthene	µg/kg lipid	yes	19.4	99.5	no
Benzo(g,h,i)perylene	µg/kg lipid	yes	6.86	99.5	no
Benzo(k)fluoranthene	µg/kg lipid	yes	11.8	99.5	no
Chrysene	µg/kg lipid	yes	41.9	99.5	no
Dibenzo(a,h)anthracene	µg/kg lipid	yes	1.35	99.5	no
Fluoranthene	µg/kg lipid	yes	233	1790	no
Fluorene	µg/kg lipid	yes	13.23	99.5	no
Indeno(1,2,3-cd)pyrene	µg/kg lipid	yes	4.69	99.5	no
Naphthalene	µg/kg lipid	yes	0.59	273.5	no
Phenanthrene	µg/kg lipid	yes	110	1190	no
Pyrene	µg/kg lipid	yes	200	99.5	**yes**
Total PAHs	µg/kg lipid	yes	526	99.5	**yes**
Phthalates					
BEHP	µg/kg lipid	no	NA	16.4	NA
Dibutyl phthalate	µg/kg lipid	yes	0.153	11.4	no
Diethyl phthalate	µg/kg lipid	no	NA	9.26	NA
Dimethyl phthalate	µg/kg lipid	no	NA	4.63	NA

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Table 3-12. Results of Round 2 COPC Predicted Tissue Screen for Laboratory-Exposed Worms

COI	Unit	Sediment-Tissue Relationship?	Maximum Predicted Tissue Concentration[a]	TRV	Round 2 COPC?[b]
SVOCs					
Dibenzofuran	µg/kg lipid	yes	13.1	42.1	no
Hexachlorobenzene	µg/kg lipid	yes	0.153	20.6	no
Hexachlorobutadiene	µg/kg lipid	yes	0.405	1.09	no
Hexachloroethane	µg/kg lipid	no	NA	1980	NA
n-Nitrosodiphenylamine	µg/kg lipid	yes	34.2	3703	no
Phenols					
2-Methylphenol	µg/kg lipid	no	NA	3220	NA
4-Methylphenol	µg/kg lipid	no	NA	3220	NA
Pentachlorophenol	µg/kg lipid	no	NA	130	NA
Phenol	µg/kg lipid	yes	2.33	261	no
Pesticides					
Aldrin	µg/kg lipid	yes	5.12	34.1	no
cis-Chlordane	µg/kg lipid	yes	1.07	23.1	no
trans-Chlordane	µg/kg lipid	yes	0.952	23.1	no
Total chlordane	µg/kg lipid	yes	3.49	23.1	no
Dieldrin	µg/kg lipid	yes	0.686	9.26	no
alpha-Endosulfan	µg/kg lipid	no	NA	3.07	NA
beta-Endosulfan	µg/kg lipid	no	NA	3.07	NA
Total endosulfan	µg/kg lipid	no	NA	3.07	NA
Endosulfan sulfate	µg/kg lipid	no	NA	3.07	NA
Endrin	µg/kg lipid	yes	2.67	1.05	**yes**
Endrin aldehyde	µg/kg lipid	no	NA	1.05	NA
Endrin ketone	µg/kg lipid	yes	0.276	1.05	no
Heptachlor	µg/kg lipid	yes	0.378	2.53	no
Heptachlor epoxide	µg/kg lipid	yes	0.0146	2.31	no
alpha-Hexachlorocyclohexane	µg/kg lipid	no	NA	0.206	NA
beta-Hexachlorocyclohexane	µg/kg lipid	yes	0.226	0.206	**yes**
delta-Hexachlorocyclohexane	µg/kg lipid	yes	0.193	0.206	no
gamma-Hexachlorocyclohexane	µg/kg lipid	no	NA	0.968	NA
Methoxychlor	µg/kg lipid	no	NA	8.42	NA
cis-Nonachlor	µg/kg lipid	yes	0.418	23.1	no
trans-Nonachlor	µg/kg lipid	yes	0.839	23.1	no
Oxychlordane	µg/kg lipid	no	NA	23.1	NA

[a] Predicted tissue concentration is based on the site-specific BSAF and the 95th percentile of detected site-wide sediment concentrations.

[b] Round 2 COPCs were identified when the predicted tissue concentration was greater than the SL TRV.

COI – chemicals of interest
COPC –chemical of potential concern
HCH –hexachlorocyclohexane
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound

Table 3-13. Results of Round 2 COPC Predicted Tissue Screen for Field-Collected Crayfish

COI	Unit	Sediment-Tissue Relationship?	Maximum Predicted Tissue Concentration[a]	TRV	Round 2 COPC?[b]
Metals					
Antimony	mg/kg ww	no	NA	0.03	NA
Arsenic	mg/kg ww	no	NA	1.7	NA
Cadmium	mg/kg ww	no	NA	0.09	NA
Chromium	mg/kg ww	no	NA	2.7	NA
Copper	mg/kg ww	no	NA	3.1	NA
Lead	mg/kg ww	yes	0.175	2.2	no
Mercury	mg/kg ww	no	NA	0.46	NA
Nickel	mg/kg ww	yes	0.571	18.4	no
Selenium	mg/kg ww	NA	NA	1.1	NA
Silver	mg/kg ww	no	NA	0.27	NA
Thallium	mg/kg ww	NA	NA	4.6	NA
Zinc	mg/kg ww	no	NA	27	NA
Butyltins					
Tributyltin ion	µg/kg lipid	NA	NA	6.22	NA
PAHs					
2-Methylnaphthalene	µg/kg lipid	NA	NA	125	NA
Acenaphthene	µg/kg lipid	NA	NA	15700	NA
Acenaphthylene	µg/kg lipid	NA	NA	125	NA
Anthracene	µg/kg lipid	NA	NA	125	NA
Benzo(a)anthracene	µg/kg lipid	yes	19.1	125	no
Benzo(a)pyrene	µg/kg lipid	NA	NA	125	NA
Benzo(b)fluoranthene	µg/kg lipid	NA	NA	125	NA
Benzo(g,h,i)perylene	µg/kg lipid	NA	NA	125	NA
Benzo(k)fluoranthene	µg/kg lipid	NA	NA	125	NA
Chrysene	µg/kg lipid	yes	15.0	125	no
Dibenzo(a,h)anthracene	µg/kg lipid	NA	NA	125	NA
Fluoranthene	µg/kg lipid	yes	29.4	2240	no
Fluorene	µg/kg lipid	NA	NA	125	NA
Indeno(1,2,3-cd)pyrene	µg/kg lipid	NA	NA	125	NA
Naphthalene	µg/kg lipid	NA	NA	810	NA
Phenanthrene	µg/kg lipid	yes	9.56	1500	no
Pyrene	µg/kg lipid	yes	37.4	125	no
Total PAHs	µg/kg lipid	yes	NA	125	NA
Phthalates					
BEHP	µg/kg lipid	NA	NA	48.6	NA
Dibutyl phthalate	µg/kg lipid	NA	NA	33.7	NA
Diethyl phthalate	µg/kg lipid	NA	NA	27.4	NA
Dimethyl phthalate	µg/kg lipid	NA	NA	13.7	NA

Table 3-13. Results of Round 2 COPC Predicted Tissue Screen for Field-Collected Crayfish

COI	Unit	Sediment-Tissue Relationship?	Maximum Predicted Tissue Concentration[a]	TRV	Round 2 COPC?[b]
SVOCs					
Dibenzofuran	µg/kg lipid	NA	NA	125	NA
Hexachlorobenzene	µg/kg lipid	NA	NA	61.1	NA
Hexachlorobutadiene	µg/kg lipid	NA	NA	3.24	NA
Hexachloroethane	µg/kg lipid	NA	NA	5860	NA
n-Nitrosodiphenylamine	µg/kg lipid	NA	NA	10970	NA
Phenols					
2-Methylphenol	µg/kg lipid	NA	NA	9540	NA
4-Methylphenol	µg/kg lipid	yes	17.39	9540	no
Pentachlorophenol	µg/kg lipid	no	NA	386	NA
Phenol	µg/kg lipid	no	NA	773	NA
Pesticides					
Aldrin	µg/kg lipid	NA	NA	101	NA
cis-Chlordane	µg/kg lipid	NA	NA	68.6	NA
trans-Chlordane	µg/kg lipid	NA	NA	68.6	NA
Total chlordane	µg/kg lipid	NA	NA	68.56	NA
Dieldrin	µg/kg lipid	NA	NA	27.4	NA
alpha-Endosulfan	µg/kg lipid	NA	NA	9.10	NA
beta-Endosulfan	µg/kg lipid	NA	NA	9.10	NA
Total Endosulfan	µg/kg lipid	NA	NA	9.10	NA
Endosulfan sulfate	µg/kg lipid	NA	NA	9.10	NA
Endrin	µg/kg lipid	NA	NA	3.12	NA
Endrin aldehyde	µg/kg lipid	NA	NA	3.12	NA
Endrin ketone	µg/kg lipid	NA	NA	3.12	NA
Heptachlor	µg/kg lipid	NA	NA	7.48	NA
Heptachlor epoxide	µg/kg lipid	NA	NA	6.86	NA
alpha-Hexachlorocyclohexane	µg/kg lipid	NA	NA	0.611	NA
beta-Hexachlorocyclohexane	µg/kg lipid	NA	NA	0.611	NA
gamma-Hexachlorocyclohexane	µg/kg lipid	NA	NA	2.87	NA
delta-Hexachlorocyclohexane	µg/kg lipid	NA	NA	0.611	NA
Methoxychlor	µg/kg lipid	NA	NA	24.9	NA
cis-Nonachlor	µg/kg lipid	NA	NA	68.6	NA
trans-Nonachlor	µg/kg lipid	NA	NA	68.6	NA
Oxychlordane	µg/kg lipid	NA	NA	68.6	NA

[a] Predicted tissue concentration is based on the site-specific BSAF and the 95th percentile of detected site-wide sediment concentrations.

[b] Round 2 COPCs were identified when the predicted tissue concentration was greater than the SL TRV.

COI – chemicals of interest
COPC –chemical of potential concern
HCH –hexachlorocyclohexane
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound

Table 3-14. Summary of Round 2 COPCs Based on the Predicted Tissue Residue Approach

Round 2 COPC	Field-Collected Clam	Laboratory-Exposed Clam	Laboratory-Exposed Worm
Metals			
Antimony			X
Arsenic			X
Cadmium	X		
Copper		X	
Zinc	X		
Butyltins			
TBT ion	X	X	X
PAHs			
Benzo(a)pyrene			X
Pyrene		X	X
Total PAHs	X	X	X
Pesticides			
beta-HCH		X	X
Endrin		X	X

COPC – chemical of potential concern
HCH –hexachlorocyclohexane
PAH – polycyclic aromatic hydrocarbon

Table 4-1. Chemicals Identified as COIs for Benthic Invertebrate Receptors Based on Near-Bottom Surface Water Data

Chemicals of Interest	
Metals	
Antimony	Nickel
Arsenic	Selenium
Cadmium	Silver
Chromium	Thallium
Copper	Zinc
Lead	
Butyltins	
Butyltin ion	Tributyltin ion
Dibutyltin ion	
PAHs	
2-Methylnaphthalene	Benzo(k)fluoranthene
Acenaphthene	Chrysene
Acenaphthylene	Dibenzo(a,h)anthracene
Anthracene	Fluoranthene
Benzo(a)anthracene	Fluorene
Benzo(a)pyrene	Indeno(1,2,3-cd)pyrene
Benzo(b)fluoranthene	Naphthalene
Benzo(b+j)fluoranthene	Phenanthrene
Benzo(g,h,i)perylene	Pyrene
Phthalates	
BEHP	Diethyl phthalate
Butylbenzyl phthalate	Dimethyl phthalate
Dibutyl phthalate	Di-n-octyl phthalate
SVOCs	
1,4-Dichlorobenzene	Hexachlorobenzene
Benzoic acid	Hexachlorobutadiene
Carbazole	Isophorone
Phenols	
4-Chloro-3-methylphenol	Phenol
PCBs	
Aroclor 1242	Aroclor 1260
Aroclor 1248	Total PCBs
Aroclor 1254	
Dioxins and Furans	
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin	1,2,3,6,7,8-Hexachlorodibenzofuran
2,3,7,8-Tetrachlorodibenzofuran	1,2,3,7,8,9-Hexachlorodibenzofuran
1,2,3,7,8-Pentachlorodibenzo-*p*-dioxin	2,3,4,6,7,8-Hexachlorodibenzofuran
2,3,4,7,8-Pentachlorodibenzofuran	1,2,3,4,6,7,8-Heptachlorodibenzo-*p*-dioxin
1,2,3,7,8-Pentachlorodibenzofuran	1,2,3,4,6,7,8-Heptachlorodibenzofuran

Table 4-1. Chemicals Identified as COIs for Benthic Invertebrate Receptors Based on Near-Bottom Surface Water Data

Chemicals of Interest	
1,2,3,4,7,8-Hexachlorodibenzo-*p*-dioxin	1,2,3,4,7,8,9-Heptachlorodibenzofuran
1,2,3,6,7,8-Hexachlorodibenzo-*p*-dioxin	Octachlorodibenzo-*p*-dioxin
1,2,3,7,8,9-Hexachlorodibenzo-*p*-dioxin	Octachlorodibenzofuran
1,2,3,4,7,8-Hexachlorodibenzofuran	
Pesticides	
2,4′-DDD	Endosulfan sulfate
2,4′-DDE	Endrin
2,4′-DDT	Endrin ketone
4,4′-DDD	Heptachlor
4,4′-DDE	Heptachlor epoxide
4,4′-DDT	alpha-HCH
Total DDTs	beta-HCH
Aldrin	delta-HCH
cis-Chlordane	gamma-HCH
trans-Chlordane	Methoxychlor
Total chlordane	cis-Nonachlor
Dieldrin	trans-Nonachlor
alpha-Endosulfan	Oxychlordane
beta-Endosulfan	
Herbicides	
Dalapon	
Perchlorate	
Perchlorate	

BEHP – bis(2-ethylhexyl) phthalate
COI – chemical of interest
HCH – hexachlorocyclohexane
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound

Table 4-2. Results of Round 2 COPC Screen of Near-Bottom Surface Water

COI	Unit	Maximum Surface Water Concentration	Chronic Eco SL	Round 2 COPC?[a]
Metals[b]				
Antimony (total)	mg/L	0.000125 T	0.03	no
Arsenic (dissolved)	mg/L	0.00064	0.15	no
Cadmium (dissolved)	mg/L	0.00003	0.00009	no
Chromium (dissolved)	mg/L	0.00033	0.0238	no
Copper (dissolved)	mg/L	0.00164 J	0.00274	no
Lead (dissolved)	mg/L	0.000152	0.00054	no
Nickel (dissolved)	mg/L	0.00112	0.0161	no
Selenium (total)	mg/L	0.0009 J	0.005	no
Silver (dissolved)	mg/L	0.000061	0.0001	no
Thallium (total)	mg/L	0.000032 JT	0.04	no
Zinc (dissolved)	mg/L	0.0419	0.0365	**yes**
Butyltins				
Butyltin ion	µg/L	0.02 J	0.072	no
Dibutyltin ion	µg/L	0.0073 J	0.072	no
Tributyltin ion	µg/L	0.0023 J	0.072	no
PAHs				
2-Methylnaphthalene	µg/L	0.22	2.1	no
Acenaphthene	µg/L	0.21	23	no
Acenaphthylene	µg/L	0.043	306.9	no
Anthracene	µg/L	0.072	0.73	no
Benzo(a)anthracene	µg/L	0.11	0.027	**yes**
Benzo(a)pyrene	µg/L	0.15	0.014	**yes**
Benzo(b)fluoranthene	µg/L	0.11	0.6774	no
Benzo(b+j)fluoranthene	µg/L	0.006768	0.6415	no
Benzo(g,h,i)perylene	µg/L	0.14	0.4391	no
Benzo(k)fluoranthene	µg/L	0.1	0.6415	no
Chrysene	µg/L	0.19	2.042	no
Dibenzo(a,h)anthracene	µg/L	0.011 J	0.2825	no
Fluoranthene	µg/L	0.4	6.16	no
Fluorene	µg/L	0.063	3.9	no
Indeno(1,2,3-cd)pyrene	µg/L	0.11	0.275	no
Naphthalene	µg/L	0.77	12	no
Phenanthrene	µg/L	0.17	6.3	no
Pyrene	µg/L	0.45	10.11	no
Phthalates				
BEHP	µg/L	0.033	3	no
Butylbenzyl phthalate	µg/L	0.027 J	3	no

Table 4-2. Results of Round 2 COPC Screen of Near-Bottom Surface Water

COI	Unit	Maximum Surface Water Concentration	Chronic Eco SL	Round 2 COPC?[a]
Dibutyl phthalate	µg/L	0.00598 J	3	no
Diethyl phthalate	µg/L	0.00665	3	no
Dimethyl phthalate	µg/L	0.00483	3	no
Di-n-octyl phthalate	µg/L	0.000142 J	3	no
SVOCs				
1,4-Dichlorobenzene	µg/L	0.019 J	15	no
Benzoic acid	µg/L	2.2 J	42	no
Carbazole	µg/L	0.06 J	18.6	no
Hexachlorobenzene	µg/L	0.00701	3.68	no
Hexachlorobutadiene	µg/L	0.00256	9.3	no
Isophorone	µg/L	0.018 J	130	no
Phenols				
4-Chloro-3-methylphenol	µg/L	0.65	0.6	**yes**
Phenol	µg/L	0.17 J	110	no
PCBs				
Aroclor 1242	µg/L	0.001083 NJ	0.053	no
Aroclor 1248	µg/L	0.00705 J	0.081	no
Aroclor 1254	µg/L	0.00657	0.033	no
Aroclor 1260	µg/L	0.008232 NJ	94	no
Total PCBs	µg/L	0.0162 J	0.014	**yes**
Dioxins and Furans				
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin[c]	pg/L	0.26	100	no
Pesticides				
2,4′-DDD	µg/L	0.002057	0.001	**yes**
2,4′-DDE	µg/L	0.0000901	0.001	no
2,4′-DDT	µg/L	0.0187 NJ	0.001	**yes**
4,4′-DDD	µg/L	0.00325	0.001	**yes**
4,4′-DDE	µg/L	0.000663	0.001	no
4,4′-DDT	µg/L	0.003862	0.001	**yes**
Total DDTs	µg/L	0.0199 NJ	0.001	**yes**
Aldrin	µg/L	0.00001627 J	0.3	no
cis-Chlordane	µg/L	0.00104 J	0.0043	no
trans-Chlordane	µg/L	0.00108 J	0.0043	no
Total chlordane	µg/L	0.00293 J	0.0043	no
Dieldrin	µg/L	0.0000625	0.056	no
alpha-Endosulfan	µg/L	0.000569	0.056	no
beta-Endosulfan	µg/L	0.0000331	0.056	no
Endosulfan sulfate	µg/L	0.00077 J	0.056	no

Table 4-2. Results of Round 2 COPC Screen of Near-Bottom Surface Water

COI	Unit	Maximum Surface Water Concentration	Chronic Eco SL	Round 2 COPC?[a]
Endrin	µg/L	0.00000157 J	0.036	no
Endrin ketone	µg/L	0.00000233 J	0.036	no
Heptachlor	µg/L	0.00000788 J	0.0038	no
Heptachlor epoxide	µg/L	0.000005262 J	0.0038	no
alpha-HCH	µg/L	0.00009525 J	0.08	no
beta-HCH	µg/L	0.00003474 J	0.08	no
gamma-HCH	µg/L	0.000768	0.08	no
delta-HCH	µg/L	0.00167 NJ	0.08	no
Methoxychlor	µg/L	0.0111	0.03	no
cis-Nonachlor	µg/L	0.00001577 J	0.0043	no
trans-Nonachlor	µg/L	0.00004879 J	0.0043	no
Oxychlordane	µg/L	0.00000267 J	0.0043	no
Herbicides				
Dalapon	µg/L	0.23	2	no
Perchlorate				
Perchlorate	µg/L	15.7 T	18	no

[a] Round 2 COPCs were identified when the maximum detected concentration in near-bottom surface water was greater than the chronic Eco SL.

[b] Concentrations of metals were evaluated as total or dissolved concentrations, depending on the Eco SL.

[c] 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin was evaluated as a representative for all dioxins and furans.

BEHP – bis(2-ethylhexyl) phthalate
COI – chemical of interest
Eco SL – Ecological Screening Level
HCH – hexachlorocyclohexane
J – estimated concentration
N – presumptive evidence of a compound
NA – not available or not applicable
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
T – value calculated or selected from multiple results
Bold identifies detected concentrations that exceed chronic Eco SLs.

Table 4-3. Chemicals Identified as COIs for Benthic Invertebrate Receptors Based on TZW Data

Chemical of Interest	
Metals	
Antimony	Nickel
Arsenic	Selenium
Barium	Silver
Cadmium	Sodium
Calcium	Thallium
Chromium	Titanium
Copper	Vanadium
Lead	Zinc
Mercury	
PAHs	
2-Methylnaphthalene	Chrysene
Acenaphthene	Dibenzo(a,h)anthracene
Acenaphthylene	Fluoranthene
Anthracene	Fluorene
Benzo(a)anthracene	Indeno(1,2,3-cd)pyrene
Benzo(a)pyrene	Naphthalene
Benzo(b)fluoranthene	Phenanthrene
Benzo(g,h,i)perylene	Pyrene
Benzo(k)fluoranthene	
SVOCs	
1,2-Dichlorobenzene	1,4-Dichlorobenzene
1,3-Dichlorobenzene	Dibenzofuran
Dioxins and Furans	
1,2,3,4,6,7,8-Heptachlorodibenzo-*p*-dioxin	1,2,3,7,8-Pentachlorodibenzofuran
1,2,3,4,7,8,9-Heptachlorodibenzofuran	2,3,4,7,8-Pentachlorodibenzofuran
1,2,3,4,7,8-Hexachlorodibenzofuran	2,3,7,8-Tetrachlorodibenzofuran
1,2,3,6,7,8-Hexachlorodibenzofuran	
Pesticides	
2,4′-DDD	4,4′-DDE
2,4′-DDT	4,4′-DDT
4,4′-DDD	Total DDTs
Herbicides	
2,4-D	Dichloroprop
Dalapon	Silvex™
VOCs	
1,1-Dichloroethane	Chloroethane
1,1-Dichloroethene	Chloroform
1,2-Dichloroethane	Chloromethane
cis-1,2-Dichloroethene	Ethylbenzene

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Table 4-3. Chemicals Identified as COIs for Benthic Invertebrate Receptors Based on TZW Data

Chemical of Interest	
trans-1,2-Dichloroethene	Isopropylbenzene
1,2-Dichloropropane	Methyl isobutyl ketone
1,1,1-Trichloroethane	Methyl n-butyl ketone
1,1,2-Trichloroethane	Methyl tert-butyl ether
1,2,4-Trimethylbenzene	Methylene bromide
1,3,5-Trimethylbenzene	Methylene chloride
1-Methyl-4-isopropylbenzene	Methylethyl ketone
Acetone	n-Propylbenzene
Acrolein	Styrene
Benzene	Tetrachloroethene
Bromochloromethane	Toluene
Bromodichloromethane	Trichloroethene
n-Butylbenzene	Vinyl chloride
sec-Butylbenzene	m,p-Xylene
tert-Butylbenzene	o-Xylene
Carbon disulfide	Total Xylenes
Chlorobenzene	
TPH[a]	
Diesel-range hydrocarbons	Residual-range hydrocarbons
Gasoline-range hydrocarbons	TPH
Cyanide	
Cyanide	
Perchlorate	
Perchlorate	
Conventionals	
Chloride	

[a] LWG and EPA are currently discussing the TPH Eco SLs and TPHs were therefore, not evaluated in the Round 2 COPC screen.

COI – chemical of interest

PAH – polycyclic aromatic hydrocarbon

SVOC – semivolatile organic compound

TPH – total petroleum hydrocarbons

VOC – volatile organic compounds

Table 4-4. Results of Round 2 COPC Screen of TZW

COI	Unit	Maximum TZW Concentration	Chronic Eco SL	Round 2 COPC?[a]
Metals[b]				
Antimony (total)	mg/L	0.0252	0.03	no
Arsenic (dissolved)	mg/L	0.0768	0.15	no
Barium (total)	mg/L	4.39	0.004	**yes**
Cadmium (dissolved)	mg/L	0.00052	0.00009	**yes**
Calcium (total)	mg/L	787 T	NA	NA
Chromium (dissolved)	mg/L	0.00891	0.0238	no
Copper (dissolved)	mg/L	0.00363	0.00274	**yes**
Lead (dissolved)	mg/L	0.00251 J	0.00054	**yes**
Mercury (dissolved)	mg/L	0.00036	0.00077	no
Nickel (dissolved)	mg/L	0.0255	0.0161	**yes**
Selenium (total)	mg/L	0.0046	0.005	no
Silver (dissolved)	mg/L	0.000048	0.0001	no
Sodium (total)	mg/L	37,490	680	**yes**
Thallium (total)	mg/L	0.000655	0.04	no
Titanium (total)	mg/L	6.97	NA	NA
Vanadium (total)	mg/L	0.379	0.02	**yes**
Zinc (dissolved)	mg/L	0.526	0.0365	**yes**
PAHs				
2-Methylnaphthalene	µg/L	84	2.1	**yes**
Acenaphthene	µg/L	399	23	**yes**
Acenaphthylene	µg/L	6.72	306.9	no
Anthracene	µg/L	63.8	0.73	**yes**
Benzo(a)anthracene	µg/L	32.3	0.027	**yes**
Benzo(a)pyrene	µg/L	37.8	0.014	**yes**
Benzo(b)fluoranthene	µg/L	33.3	0.6774	**yes**
Benzo(g,h,i)perylene	µg/L	28.8	0.4391	**yes**
Benzo(k)fluoranthene	µg/L	9	0.6415	**yes**
Chrysene	µg/L	34.5	2.042	**yes**
Dibenzo(a,h)anthracene	µg/L	3.71	0.2825	**yes**
Fluoranthene	µg/L	106	6.16	**yes**
Fluorene	µg/L	108	3.9	**yes**
Indeno(1,2,3-cd)pyrene	µg/L	16.9	0.275	**yes**
Naphthalene	µg/L	13,700	12	**yes**
Phenanthrene	µg/L	362	6.3	**yes**
Pyrene	µg/L	148	10.11	**yes**
SVOCs				
1,2-Dichlorobenzene	µg/L	640	14	**yes**
1,3-Dichlorobenzene	µg/L	23	71	no

Table 4-4. Results of Round 2 COPC Screen of TZW

COI	Unit	Maximum TZW Concentration	Chronic Eco SL	Round 2 COPC?[a]
1,4-Dichlorobenzene	µg/L	240	15	**yes**
Dibenzofuran	µg/L	8	3.7	**yes**
Pesticides				
2,4'-DDD	µg/L	1.1 J	0.001	**yes**
2,4'-DDT	µg/L	0.093 J	0.001	**yes**
4,4'-DDD	µg/L	1.3	0.001	**yes**
4,4'-DDE	µg/L	0.12 J	0.001	**yes**
4,4'-DDT	µg/L	1.8	0.001	**yes**
Total DDTs	µg/L	3.05 J	0.001	**yes**
Herbicides				
2,4-D	µg/L	0.97 J	4	no
Dalapon	µg/L	2.4	2	**yes**
Dichloroprop	µg/L	0.7 J	10	no
Silvex™	µg/L	22	5	**yes**
VOCs				
1,1,1-Trichloroethane	µg/L	0.33 J	11	no
1,1,2-Trichloroethane	µg/L	7.65	9400	no
1,1-Dichloroethane	µg/L	27	47	no
1,1-Dichloroethene	µg/L	40.5	25	**yes**
1,2,4-Trimethylbenzene	µg/L	69.9	7.3	**yes**
1,2-Dichloroethane	µg/L	2.3	910	no
1,2-Dichloropropane	µg/L	0.32 J	5700	no
1,3,5-Trimethylbenzene	µg/L	21.6	7.3	**yes**
1-Methyl-4-isopropylbenzene	µg/L	3.98	9.8	no
Acetone	µg/L	34 J	1500	no
Acrolein	µg/L	2.4 J	21	no
Benzene	µg/L	3,840	130	**yes**
Bromochloromethane	µg/L	0.22 J	220	no
Bromodichloromethane	µg/L	0.31 J	220	no
Carbon disulfide	µg/L	800	0.92	**yes**
Chlorobenzene	µg/L	12,000	50	**yes**
Chloroethane	µg/L	160	47	**yes**
Chloroform	µg/L	580	1240	no
Chloromethane	µg/L	11	220	no
cis-1,2-Dichloroethene	µg/L	67,000	590	**yes**
Ethylbenzene	µg/L	416	7.3	**yes**
Isopropylbenzene	µg/L	14.5	7.3	**yes**
m,p-Xylene	µg/L	293	66.67	**yes**
Methyl isobutyl ketone	µg/L	5.5 J	170	no
Methyl N-butyl ketone	µg/L	13 J	99	no

Table 4-4. Results of Round 2 COPC Screen of TZW

COI	Unit	Maximum TZW Concentration	Chronic Eco SL	Round 2 COPC?[a]
Methyl tert-butyl ether	µg/L	4.4	10000	no
Methylene bromide	µg/L	0.21 J	11	no
Methylene chloride	µg/L	2	2200	no
Methylethyl ketone	µg/L	12 J	14000	no
n-Butylbenzene	µg/L	0.48	7.3	no
n-Propylbenzene	µg/L	3.95	7.3	no
o-Xylene	µg/L	150	13	**yes**
Sec-butylbenzene	µg/L	7.06	7.3	no
Styrene	µg/L	1.71	4	no
tert-Butylbenzene	µg/L	0.4	7.3	no
Tetrachloroethene	µg/L	50	840	no
Toluene	µg/L	178	9.8	**yes**
trans-1,2-Dichloroethene	µg/L	117	590	no
Trichloroethene	µg/L	88,500	21900	**yes**
Vinyl chloride	µg/L	4,300	3880	**yes**
Total xylenes	µg/L	440	13	**yes**
TPH				
Diesel Range Hydrocarbons	µg/L	6,100 J	NA	NA
Gasoline Range Hydrocarbons	µg/L	4,000 J	NA	NA
Residual Range Hydrocarbons	µg/L	1,200 J	NA	NA
Total Petroleum Hydrocarbons	µg/L	11,300 J	NA	NA
Cyanide				
Cyanide	mg/L	23.1 J	0.0052	**yes**
Perchlorate				
Perchlorate	µg/L	177,000	18	**yes**
Conventionals				
Chloride	mg/L	51,700	NA	NA

[a] Round 2 COPCs were identified when the maximum detected concentration in TZW was greater than the chronic Eco SL.

[b] Concentrations of metals were evaluated as total or dissolved concentrations, depending on the Eco SL.

BEHP – bis(2-ethylhexyl) phthalate
COI – chemical of interest
Eco SL – Ecological Screening Level
HCH – hexachlorocyclohexane
J – estimated concentration
NA – not available or not applicable
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
VOC – volatile organic compound
T – value calculated or selected from multiple results
TPH – total petroleum hydrocarbons

Bold identifies TZW concentrations that exceed chronic Eco SLs.

Table 4-5. Results of Round 2 COPC Screen of EqP-Predicted TZW Concentrations

COI	Maximum Sediment Concentration (mg/kg-OC)	K_{oc}	Maximum Predicted TZW Concentration[a] (µg/L)	Chronic Eco SL (µg/L)	Round 2 COPC[b]?
PAHs					
Acenaphthylene	1,600	4074	390	306.9	**yes**
VOCs					
1,1-Dichloroethane	0.025	57.54	0.43	47	no
1,2-Dichloroethane	0.027	28.18	0.96	910	no
Acetone	10	5.37	1,900	1,500	**yes**
Chloroform	4.8	105.9	45	1,240	no
Methylene chloride	0.65	25.12	26	2,200	no
trans-1,2-Dichloroethene	0.072	36.3	2.0	590	no

[a] TZW concentrations were predicted in the EqP approach by dividing the maximum OC-normalized sediment concentrations by the K_{OC} values.

[b] Round 2 COPCs were identified when the maximum predicted TZW concentration was greater than the chronic Eco SL.

Eco SL – ecological screening level

EqP – equilibrium partitioning

PAH – polycyclic aromatic hydrocarbon

TZW – transition zone water

VOC – volatile organic compound

Bold identifies predicted TZW concentrations that exceed Eco SLs.

FIGURES

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Figure 2-1
Predicted Benthic Toxicity Based on TEC/PEC
River Miles 2 to 11

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Comprehensive Round 2 Report
Attachment G2

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Figure 2-2
Predicted Benthic Toxicity Based on TEL/PEL
River Miles 2 to 11

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Comprehensive Round 2 Report
Attachment G2

Prepared by STS, 11/09/06. Modified by AMW, 01/30/07. Map 2939. W:\Projects\01-25-01 Portland Harbor ERA\Data\GIS\Events\BenthicRisk\FPM_SQV_Toxicity

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Figure 2-3
Predicted Benthic Toxicity Based on LEL/SEL
River Miles 2 to 11

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Attachment G2

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Figure 2-4
Predicted Benthic Toxicity Based on SQS/CSL
River Miles 2 to 11

Portland Harbor RI/FS
Comprehensive Round 2 Report
Attachment G2

Prepared by STS, 11/16/06. Modified by AMW, 01/30/07. Map 7255. W:\Projects\01-2501 Portland Harbor ERA\Data\GIS\Events\Benthic.mxd

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Figure 2-5
Predicted Benthic Toxicity Based on ERM Quotient
River Miles 2 to 11

Portland Harbor RI/FS
Comprehensive Round 2 Report
Attachment G2

APPENDIX G: ECOLOGICAL RISK ASSESSMENT

Attachment G3
Ecological Screening Levels

ECOLOGICAL SCREENING LEVELS

A review of water quality regulatory benchmarks and literature-based thresholds was conducted to develop Ecological Screening Levels (Eco SLs) for chemicals in water. These Eco SLs were developed to evaluate potential risks to aquatic receptors (i.e., benthic invertebrates, fish, amphibians, and aquatic plants) exposed to chemical concentrations in water in the Round 2 ERA. The process for developing Eco SLs is presented in the Technical Memorandum, *Process for selecting acute and chronic water screening levels for Portland Harbor surface water, groundwater, and transition zone water* (Windward 2005). Eco SLs were developed by LWG and revised based on EPA's comments (EPA 2006a). In general, EPA believes that the selected Eco SLs should be protective of all aquatic receptors (EPA 2006a); however, EPA and LWG are still discussing the application of the water quality criteria hierarchy for the Baseline Ecological Risk Assessment (LWG 2006).

LWG adopted the following hierarchy proposed by EPA to select Eco SLs for identifying surface water and transition zone water (TZW) Round 2 COPCs:

- **Level 1** – The lower of the national recommended water quality criteria (i.e., ambient water quality criteria [AWQC]) and the proposed State of Oregon water quality criteria as specified in OAR 340-41 Table 33.
- **Level 2** – The Tier II values from Suter and Tsao (1996).
- **Level 3** – EPA-proposed polycyclic aromatic hydrocarbons (PAH)-specific final chronic values for individual PAH compounds (Table 3-4 of EPA 2003b). Use of these individual PAH guidelines as screening levels eliminates the need to use benzo(a)pyrene as a surrogate for other PAH compounds.
- **Level 4** – Canadian water environmental quality guidelines.
- **Level 5** –Either acute ODEQ guidance values (ODEQ 2006) or Tier II acute values for Suter and Tsao (1996) divided by a UF of 50, whichever is more protective.
- **Level 6** – Literature-derived values.

Table 1 presents the Eco SLs used in the Round 2 ERA for all surface water and TZW chemicals of interest (COIs). COIs were defined in the Round 2 ERA as those detected chemicals in surface water and or TZW samples (see Appendix G).

TABLES

Table 1. Selected Acute and Chronic Eco SLs for Chemicals in Water

Chemical of Interest[a]	Selected Acute Eco SL (µg/L)	Selected Chronic Eco SL (µg/L)	Eco SL Source[b]	Comments
Metals				
Antimony	180	30	Tier II	Eco SLs are based on total.
Arsenic	340	150	AWQC	Eco SLs are based on dissolved fraction for As (III).
Barium	110	4	Tier II	Eco SLs are based on total.
Cadmium	0.52	0.09	AWQC	Eco SLs are hardness-dependent and were adjusted (see Appendix G) using AWQC equations to correspond with a hardness of 25 mg/L calcium carbonate (estimate for the Lower Willamette River [LWR]). Eco SLs are based on dissolved fraction.
Calcium	NA	NA	NA	No toxicity data were available.
Chromium (III)	183.1	23.8	AWQC	Eco SLs are hardness-dependent and were adjusted (see Appendix G) using AWQC equations to correspond with a hardness of 25 mg/L calcium carbonate (estimate for the LWR). Eco SLs are based on dissolved fraction.
Copper	3.64	2.74	AWQC	Eco SLs are hardness-dependent and were adjusted (see Appendix G) using AWQC equations to correspond with a hardness of 25 mg/L calcium carbonate (estimate for the LWR). Eco SLs are based on dissolved fraction.
Lead	13.88	0.54	AWQC	Eco SLs are hardness-dependent and were adjusted (see Appendix G) using AWQC equations to correspond with a hardness of 25 mg/L calcium carbonate (estimate for the LWR). Eco SLs are based on dissolved fraction.
Mercury	1.4	0.77	AWQC	Criteria are for dissolved fraction.
Nickel	144.9	16.1	AWQC	Eco SLs are hardness-dependent and were adjusted (see Appendix G) using AWQC equations to correspond with a hardness of 25 mg/L calcium carbonate (estimate for the LWR). Eco SLs are based on dissolved fraction.
Selenium	12.8	5	AWQC	The listed acute value is for selenate, and is the lower of the published criteria for selenate and selenite. Criteria are for total.
Silver	0.3	0.1	AWQC (acute); ODEQ (chronic)	Eco SLs are hardness-dependent and were adjusted (see Appendix G) using AWQC equations to correspond with a hardness of 25 mg/L calcium carbonate (estimate for the LWR). Eco SLs are based on dissolved fraction. ODEQ guidance value is based on old AWQC (EPA 1980).

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Table 1. Selected Acute and Chronic Eco SLs for Chemicals in Water

Chemical of Interest[a]	Selected Acute Eco SL (µg/L)	Selected Chronic Eco SL (µg/L)	Eco SL Source[b]	Comments
Sodium	6,800,000	680,000	LCV	Eco SLs are based on total. Acute Eco SL was derived by multiplying the chronic Eco SL by 10.
Thallium	1,400	40	ODEQ	Eco SLs are based on total.
Titanium	NA	NA	NA	No toxicity data were available.
Vanadium	280	20	Tier II	Eco SLs are based on total.
Zinc	36.2	36.5	AWQC	Eco SLs are hardness-dependent and were adjusted (see Appendix G) using AWQC equations to correspond with a hardness of 25 mg/L calcium carbonate (estimate for the LWR). Eco SLs are based on dissolved fraction.
Butyltins				
Butyltin ion	0.46	0.072	AWQC	Eco SLs for TBT were used as a surrogate.
Dibutyltin ion	0.46	0.072	AWQC	Eco SLs for TBT used as a surrogate.
Tributyltin (TBT) ion	0.46	0.072	AWQC	AWQC apply to TBT ion concentration (EPA 2003a)
PAHs				
2-Methylnaphthalene	37	2.1	Tier II	The Eco SLs for 1-methynapthalene were used as a surrogate.
Acenaphthene	80	23	Tier II	The acute and chronic Eco SLs are the EPA calculated final acute value (FAV) and final chronic value (FCV), respectively, for sediment quality guideline development.
Acenaphthylene	NA	306.9	EPA (2003b)	Per EPA (EPA 2006a), the acute Eco SL was not adopted.
Anthracene	13	0.73	Tier II	Eco SLs were calculated for AWQC, though were never adopted.
Benzo(a)anthracene	0.49	0.027	Tier II	
Benzo(a)pyrene	0.24	0.014	Tier II	
Benzo(b)fluoranthene	NA	0.6774	EPA (2003b)	Per EPA (EPA 2006a), the acute Eco SL was not adopted.
Benzo(b+j)fluoranthene	NA	0.6415	EPA (2003b)	Benzo(k)fluoranthene used as surrogate for the chronic Eco SL. Per EPA (EPA 2006a), the acute Eco SL was not adopted.
Benzo(g,h,i)perylene	NA	0.4391	EPA (2003b)	Per EPA (EPA 2006a), the acute Eco SL was not adopted.
Benzo(k)fluoranthene	NA	0.6415	EPA (2003b)	Per EPA (EPA 2006a), the acute Eco SL was not adopted.
Chrysene	NA	2.042	EPA (2003b)	Per EPA (EPA 2006a), the acute Eco SL was not adopted.

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Table 1. Selected Acute and Chronic Eco SLs for Chemicals in Water

Chemical of Interest[a]	Selected Acute Eco SL (µg/L)	Selected Chronic Eco SL (µg/L)	Eco SL Source[b]	Comments
Dibenz(a,h)anthracene	NA	0.2825	EPA (2003b)	Per EPA (EPA 2006a), the acute Eco SL was not adopted.
Fluoranthene	33.6	6.16	Tier II	Eco SLs were calculated for AWQC, though were never adopted.
Fluorene	70	3.9	Tier II	The acute and chronic Eco SLs are the EPA calculated FAV and FCV, respectively, for sediment quality guideline development.
Indeno(1,2,3-cd)pyrene	NA	0.275	EPA (2003b)	Per EPA (EPA 2006a), the acute Eco SL was not adopted.
Naphthalene	190	12	Tier II	
Phenanthrene	30	6.3	Tier II	Eco SLs were calculated for AWQC, though were never adopted.
Pyrene	NA	10.11	EPA (2003b)	Per EPA (EPA 2006a), the acute Eco SL was not adopted.
Phthalates				
BEHP	27	3	Tier II	
Butylbenzyl phthalate	150	3	Tier II	Per EPA (EPA 2006a), the chronic Eco SL for BEHP was used a surrogate and the acute Eco SL was derived by multiplying the chronic Eco SL by 50.
Dibutyl phthalate	150	3	Tier II	Per EPA (EPA 2006a), the chronic Eco SL for BEHP was used a surrogate and the acute Eco SL was derived by multiplying the chronic Eco SL by 50.
Diethyl phthalate	150	3	Tier II	Per EPA (EPA 2006a), the chronic Eco SL for BEHP was used a surrogate and the acute Eco SL was derived by multiplying the chronic Eco SL by 50.
Dimethyl phthalate	150	3	Tier II	Per EPA (EPA 2006a), the chronic Eco SL for BEHP was used a surrogate and the acute Eco SL was derived by multiplying the chronic Eco SL by 50.
Di-n-octyl phthalate	150	3	Tier II	Per EPA (EPA 2006a), the chronic Eco SL for BEHP was used a surrogate and the acute Eco SL was derived by multiplying the chronic Eco SL by 50.
SVOCs				
1,2-Dichlorobenzene	260	14	Tier II	
1,3-Dichlorobenzene	630	71	Tier II	
1,4-Dichlorobenzene	180	15	Tier II	
Benzoic acid	740	42	Tier II	
Dibenzofuran	66	3.7	Tier II	

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Table 1. Selected Acute and Chronic Eco SLs for Chemicals in Water

Chemical of Interest[a]	Selected Acute Eco SL (µg/L)	Selected Chronic Eco SL (µg/L)	Eco SL Source[b]	Comments
Hexachlorobenzene	6.0	3.68	EPA (2006a) (acute); MacDonald ES (1999) (chronic)	Per EPA (EPA 2006a), acute Eco SL is EPA draft criterion.
Hexachlorobutadiene	90	9.3	ODEQ	
Phenols				
4-chloro-3-methylphenol	30	0.60	ODEQ	Chronic Eco SL was derived by dividing the acute Eco SL by 50.
Phenol	3600	110	Tier II	Eco SLs were calculated for AWQC, though were never adopted.
PCBs				
Aroclor 1242	1.2	0.053	Tier II	
Aroclor 1248	1.4	0.081	Tier II	
Aroclor 1254	0.6	0.033	Tier II	
Aroclor 1260	1,700	94	Tier II	
Total PCBs	2	0.014	ODEQ (acute); Tier II (chronic)	Eco SLs were calculated for AWQC, though were never adopted. Eco SLs for total PCBs applies to the sum of all homologs, sum of Aroclors, or sum of congeners.
Dioxins and Furans				
2,3,7,8 TCDD	0.01	0.0001	ODEQ (acute); AWQC (EPA 1984) (chronic)	Eco SLs for 2,3,7,8-TCDD were evaluated as a representative for all other individual dioxins and furan.
Pesticides				
Aldrin	3	0.3	AWQC (acute); MacDonald ES (1999) (chronic)	
cis-Chlordane	1.2	0.0043	AWQC	Per EPA (EPA 2006a), Eco SLs for chlordane was used as a surrogate.
trans-Chlordane	1.2	0.0043	AWQC	Per EPA (EPA 2006a), Eco SLs for chlordane was used as a surrogate.
cis-Nonachlor	1.2	0.0043	AWQC	Per EPA (EPA 2006a), Eco SLs for chlordane was used as a surrogate.
trans-Nonachlor	1.2	0.0043	AWQC	Eco SLs for chlordane was used as a surrogate.
Oxychlordane	1.2	0.0043	AWQC	Per EPA (EPA 2006a), Eco SLs for chlordane were used as a surrogate.

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Table 1. Selected Acute and Chronic Eco SLs for Chemicals in Water

Chemical of Interest[a]	Selected Acute Eco SL (µg/L)	Selected Chronic Eco SL (µg/L)	Eco SL Source[b]	Comments
Total chlordane	1.2	0.0043	AWQC	Eco SLs for chlordane was used as a surrogate. Per EPA (EPA 2006a), acute AWQC was derived by dividing by 2 to derive the acute Eco SL.
2,4'-DDD	1.1	0.001	AWQC	AWQC for 4,4'-DDT applies to 2,4'-DDD.
2,4'-DDE	1.1	0.001	AWQC	AWQC for 4,4'-DDT applies to 2,4'-DDE.
2,4'-DDT	1.1	0.001	AWQC	AWQC for 4,4'-DDT applies to 2,4'-DDT.
4,4'-DDD	1.1	0.001	AWQC	AWQC for 4,4'-DDT applies to 4,4'-DDD.
4,4'-DDE	1.1	0.001	AWQC	AWQC for 4,4'-DDT applies to 4,4'-DDE.
4,4'-DDT	1.1	0.001	AWQC	
Total DDT	1.1	0.001	AWQC	AWQC for 4,4'-DDT applies to total DDTs.
Dieldrin	0.24	0.056	AWQC	
alpha-Endosulfan	0.22	0.056	AWQC	AWQC were derived for total endosulfan were applied to alpha- and beta-endosulfan individually.
beta-Endosulfan	0.22	0.056	AWQC	AWQC were derived for total endosulfan were applied to alpha- and beta-endosulfan individually.
Endosulfan sulfate	0.22	0.056	AWQC	Eco SLs for alpha- and beta-endosulfan were used as a surrogate.
Endrin	0.086	0.036	AWQC	
Endrin ketone	0.086	0.036	AWQC	Eco SLs for endrin were used as a surrogate.
Heptachlor	0.26	0.0038	AWQC	Acute Eco SL was derived by dividing the acute AWQC by 2.
Heptachlor epoxide	0.26	0.0038	AWQC	Eco SLs based on heptachlor, which may also be applied to heptachlor epoxide. Acute Eco SL was derived by dividing the acute AWQC by 2.
alpha-HCH	0.95	0.08	AWQC (acute); ODEQ (chronic)	Eco SLs for gamma-HCH were used as a surrogate.
beta-HCH	0.95	0.08	AWQC (acute); ODEQ (chronic)	Eco SLs for gamma-HCH were used as a surrogate.
gamma-HCH	0.95	0.08	AWQC (acute); ODEQ (chronic)	

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Table 1. Selected Acute and Chronic Eco SLs for Chemicals in Water

Chemical of Interest[a]	Selected Acute Eco SL (µg/L)	Selected Chronic Eco SL (µg/L)	Eco SL Source[b]	Comments
delta-HCH	0.95	0.08	AWQC (acute); ODEQ (chronic)	Per EPA (EPA 2006a), Eco SLs for gamma-HCH were used as a surrogate; however, the use of gamma-HCH as a surrogate requires more technical explanation.
Methoxychlor	0.3	0.03	AWQC	Acute Eco SL was derived by multiplying the chronic Eco SL by 10.
Herbicides				
2,4' D	800	4	ECOTOX (acute); MacDonald ES (1999) (chronic)	Acute Eco SL was derived by dividing the LC50 reported in ECOTOX by 2. Chronic Eco SL is the Ontario Water Quality objective.
Carbazole	465	18.6	Brooke (1991)	Chronic and acute Eco SLs were derived by dividing the LC50 reported by Brooke (1991) by 50 and 2, respectively.
Dalapon	NA	2	George et al. (1982)	
Dichloroprop	250	10	EPA (2000)	Chronic and acute Eco SLs were derived by dividing the LC50 reported by EPA (2000) by 50 and 2, respectively.
Silvex™	125	5.0	ECOTOX	Chronic and acute Eco SLs were derived by dividing the LC50 reported by ECOTOX by 50 and 2, respectively.
VOCs				
1-methyl-4-isopropylbenzene (p-Isopropyltoluene)	17,500	9.8	ODEQ (acute); Tier II (chronic)	Eco SLs for toluene were used as a surrogate.
1,1,1-Trichloroethane	200	11	Tier II	
1,1,2-Trichloroethane	18,000	9,400	ODEQ	
1,1-Dichloroethane	830	47	Tier II	
1,1-Dichloroethene	450	25	Tier II	
cis-1,2-Dichloroethene	11,600	590	ODEQ (acute); Tier II (chronic)	The acute Eco SL for dichloroethylenes was applied to both cis- and trans-congeners.
trans-1,2-Dichloroethene	11,600	590	ODEQ (acute); Tier II (chronic)	The acute Eco SL for dichloroethylenes was applied to both cis- and trans-congeners.
1,2-Dichloroethane	8,800	910	Tier II	
1,2-Dichloropropane	23,000	5,700	ODEQ	

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Table 1. Selected Acute and Chronic Eco SLs for Chemicals in Water

Chemical of Interest[a]	Selected Acute Eco SL (µg/L)	Selected Chronic Eco SL (µg/L)	Eco SL Source[b]	Comments
1,2,4-Trimethylbenzene	130	7.3	ODEQ (acute); Tier II (chronic)	Eco SLs for ethylbenzene were used as a surrogate.
1,3,5-Trimethylbenzene	130	7.3	ODEQ (acute); Tier II (chronic)	Eco SLs for ethylbenzene were used as a surrogate.
Acetone	28,000	1,500	Tier II	
Acrolein	68	21	ODEQ	
Benzene	2,300	130	Tier II	
Bromodichloromethane	11,000	220	ODEQ	Eco SLs based on the ODEQ criteria for halomethanes. Chronic Eco SL was derived by dividing the acute Eco SL by 50.
Bromomethane	11,000	220	ODEQ	Eco SLs based on the ODEQ criteria for halomethanes. Chronic Eco SL was derived by dividing the acute Eco SL by 50.
n-Butylbenzene	130	7.3	ODEQ (acute); Tier II (chronic)	Eco SLs for ethylbenzene were used as a surrogate.
sec-Butylbenzene	130	7.3	ODEQ (acute); Tier II (chronic)	Eco SLs for ethylbenzene were used as a surrogate.
tert-Butylbenzene	130	7.3	ODEQ (acute); Tier II (chronic)	Eco SLs for ethylbenzene were used as a surrogate.
Carbon disulfide	17	0.92	Tier II	
Chlorobenzene	250	50	ODEQ	The Eco SLs are for chlorobenzenes, not chlorobenzene (i.e. monochlorobenzene) specifically.
Chloroethane	830	47	Tier II	Eco SLs for 1,1-dichloroethane were used as a surrogate.
Chloroform	28,900	1,240	ODEQ	
Chloromethane	11,000	220	ODEQ	Eco SLs based on the ODEQ criteria for halomethanes. Chronic Eco SL was derived by dividing the acute Eco SL by 50.
Ethylbenzene	32,000	7.3	ODEQ (acute); Tier II (chronic)	
Isophorone	117,000	130	ODEQ (acute); MacDonald ES (1999) (chronic)	

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Table 1. Selected Acute and Chronic Eco SLs for Chemicals in Water

Chemical of Interest[a]	Selected Acute Eco SL (µg/L)	Selected Chronic Eco SL (µg/L)	Eco SL Source[b]	Comments
Isopropylbenzene (1-Methylethylbenzene)	130	7.3	ODEQ (acute); Tier II (chronic)	Eco SLs for ethylbenzene were used as a surrogate.
Methyl isobutyl ketone	2,200	170	Tier II	
Methyl n-butyl ketone (2-Hexanone)	1,800	99	Tier II	
Methyl tert-butyl ether (MTBE)	151,000	10,000	Mancini et al. (2002) (acute); Canadian EQG (chronic)	
Methylene bromide	11,000	11	ODEQ (acute); MacDonald ES (1999) (chronic)	The acute Eco SL is based on the ODEQ criteria for halomethanes.
Methylene chloride	11,000	2,200	ODEQ (acute); Tier II (chronic)	The acute Eco SL is based on the ODEQ criteria for halomethanes.
Methylethyl ketone	240,000	14,000	Tier II	
n-Propylbenzene	130	7.3	ODEQ (acute); Tier II (chronic)	Eco SLs for ethylbenzene were used as a surrogate.
Styrene	NA	4	MacDonald ES (1999)	
Tetrachloroethene	5,280	840	ODEQ	
Toluene	17,500	9.8	ODEQ (acute); Tier II (chronic)	
Trichloroethene	45,000	21,900	ODEQ	
Trichlorofluoromethane	11,000	580	ODEQ (acute); MacDonald ES (1999) (chronic)	The acute Eco SL is based on the ODEQ criteria for halomethanes.
Vinyl chloride	97,000	3,880	Brown et al. (1977)	Chronic and acute Eco SLs were derived by dividing the LC100 reported by Brown et al. (1977) by 50 and 2, respectively.
m,p-Xylene	1,200	66.67	EPA (2006b)	Eco SLs for m-xylene were used.
o-Xylene	230	13	Tier II	Eco SLs for xylene were used as a surrogate.
Total Xylene	230	13	Tier II	Eco SLs for xylene were used as a surrogate.

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Table 1. Selected Acute and Chronic Eco SLs for Chemicals in Water

Chemical of Interest[a]	Selected Acute Eco SL (µg/L)	Selected Chronic Eco SL (µg/L)	Eco SL Source[b]	Comments
TPH				
Diesel-range, gasoline-range, residual-range hydrocarbons, and TPH	NA	NA	NA	Per EPA (EPA 2006a), the EPA proposed Eco SL (0.014 ug/l) for TPHs is pending further technical justification.) The proposed Eco SL is a narrative water quality number and does not meet the data acceptability criteria of a chronic endpoint based on survival, growth, reproduction.
Cyanide				
Cyanide	22	5.2	AWQC	
Perchlorate				
Perchlorate	NA	18	Goleman et al. (2002)	Chronic Eco SL based on amphibian growth LOAEL reported by Goleman et al. (2002).
Conventionals				
Chloride	NA	NA	NA	No toxicity data were available.

[a] Contaminants of interest (COIs) were defined in the Round 2 ERA as those detected chemicals in surface water and or TZW samples (see Appendix G).

[b] Ambient Water Quality Criteria (AWQC) are from EPA (2002) and EPA (2006c). Tier II and lowest chronic values (LCV) are from Suter and Tsao (1996). Oregon Department of Environmental Quality (ODEQ) values are from ODEQ (2006). PAH mixture values are from EPA (2003b). Canadian EQG are from Canadian Water Quality Guidelines updated 2005 (CCME 2005). MacDonald ES (1999) are from Environment Canada, Georgia Basin Action Plan compendium of environmental quality benchmarks. ECOTOX values were based on EPA's online toxicity database (ECOTOX 2006).

AWQC – ambient water quality criteria
BEHP – Bis(2-ethylhexyl) phthalate
Eco SL – ecological screening level
EPA – US environmental protection agency
HCH – hexachlorocyclohexane
LCV – lowest chronic value
LC50 – dose that is lethal to 50% of an exposed population
LC100 – dose that is lethal to 100% of an exposed population
NA – not available
ODEQ – Oregon Department of Environmental Quality
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCP – pentachlorophenol
SL – screening level
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TPH – total petroleum hydrocarbons

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APPENDIX G: ECOLOGICAL RISK ASSESSMENT

Attachment G4
Screening Assessment
for Fish

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1.0 INITIAL SCREENING PROCESS

In the initial screening process for fish receptors, all lines of evidence associated with the four measures of exposure and effects were evaluated (see Section 4.0 of Appendix G). These four measures were: 1) empirical and predicted fish chemistry tissue data compared to tissue residue toxicity reference values (TRVs), 2) dietary dose estimates compared to dietary dose TRVs, 3) water data compared to ambient water quality criteria (AWQC) or TRVs and compared to concentrations in water associated with olfactory function impairment in migrating adult chinook salmon populations, and 4) PAH sediment chemistry data compared to literature-derived polycyclic aromatic hydrocarbon (PAH) concentrations associated with the occurrence of skin or liver lesions in fish.

These four measures of exposure and effects included an initial screening process, which is presented in the following sections: evaluation of empirical tissue chemistry data (Section 2), evaluation of predicted sculpin tissue chemistry data (Section 3), evaluation of dietary dose estimates (Section 4), and evaluation of water data (Section 5), and evaluation of PAH sediment data (Section 6).

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2.0 TISSUE RESIDUE ASSESSMENT

The critical tissue residue assessment was one line of evidence for evaluating risks to selected fish receptors. In this assessment, whole-body tissue concentrations of ecological fish receptor species (i.e., largescale sucker, carp, sculpin, peamouth, juvenile chinook salmon, smallmouth bass, and northern pikeminnow) were compared to tissue TRVs for each chemical and fish receptor pair.

This section presents the initial screening process for the tissue residue assessment, including the tissue-residue screen and chemicals of potential concern (COPCs) identification. These COPCs were evaluated in the fish risk assessment (see Section 4.0 of Appendix G).

2.1 TISSUE SCREEN

To identify COPCs for fish using the critical tissue residue approach, the maximum whole-body tissue concentration[1] of each chemical of interest (COI) was compared to a screening level (SL) TRV for that chemical. If the maximum exposure concentration (for each receptor) was greater than the SL TRV, the chemical was identified as a COPC and carried forward in the risk evaluation. The identification of COPCs by means of a conservative screen in which maximum concentrations in relevant media are used is consistent with US Environmental Protection Agency (EPA) guidance (EPA 1997a, b) and an important step for narrowing the list of chemicals to only those chemicals that could potentially pose a risk to selected fish receptors.

2.2 SELECTION OF COIS

COIs for fish were defined as all chemicals detected in whole-body fish tissue from the ecological risk assessment (ERA) dataset (Table 2-1), excluding crustal elements (i.e., aluminum and manganese; see Section 2.0 of Appendix G).

All fish COIs, with the exception of PAHs, which are extensively metabolized by fish, were retained for screening level evaluation to identify Round 2 COPCs using the critical tissue residue approach. PAHs were evaluated in the dietary line of evidence (Section 3), surface water line of evidence (Section 4), and PAH sediment exposure line of evidence (Section 5).

Evaluation of dioxins and furans was limited to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD). A toxic equivalent (TEQ) approach can be used to evaluate the risks associated with other dioxins and furans, as well as dioxin-like PCB congeners, based on fish TEFs developed by the World Health Organization (WHO) in 1998 (Van den Berg et al. 1998). However, because of the high uncertainty associated with fish

[1] Maximum concentrations were defined as the higher of the maximum detected concentration or the maximum reporting limit (RL) of a non-detected concentration

TEFs, PCB TEQ and dioxin TEQ values were not calculated, and 2,3,7,8-TCDD was the only dioxin or furan evaluated. PCB TEQ and dioxin TEQ tissue concentrations for fish are presented in the uncertainty analysis of Section 4.0.

2.3 ROUND 2 COPC IDENTIFICATION

Round 2 COPCs were identified for each fish receptor by comparing maximum whole-body tissue concentrations reported (either detected or non-detected) for all COIs to SL TRVs. If the maximum tissue concentration of a receptor was greater than the SL TRV, the chemical and receptor were identified as a COPC/receptor pair for this Round 2 ERA.

The approach for developing aquatic tissue residue SL TRVs were developed by EPA and its partners (EPA 2005) for data evaluation in the ecological preliminary risk evaluation (Ecological PRE) (Windward 2005). In this conservative approach, a single TRV was developed for each chemical to be protective of all aquatic species (i.e., representative of all aquatic receptors, either fish or invertebrate), rather than multiple TRVs (i.e., NOAELs and LOAELs) for each receptor group. The aquatic tissue residue SL TRVs were developed using the two-tiered approach as follows:

- **Tier 1** – For those COIs for which a sufficient number of studies (i.e., at least 20) were included in Appendix B of the Ecological PRE (Windward 2005), the SL TRV was represented as the fifth percentile of all aquatic species LOAEL data (including fish, crayfish, and clam). This approach (using species sensitivity distribution), was intended to define a critical tissue residue concentration that is protective of 95% of aquatic species. The TRV technical memorandum in Appendix B of the Ecological PRE presents the results of the entire literature search process and a detailed discussion of how each TRV was derived from the reviewed studies.

  For those COIs for which fewer than 20 studies were identified in Appendix B of the Ecological PRE, the fifth percentile tissue screening concentration reported in Dyer et al. (2000) was selected as the TRV. Literature-derived values included in Dyer et al. (2000) were based on single chemical laboratory tests that reported whole-body tissue concentrations associated with adverse effects on survival, reproduction, growth, behavior, or morphology. Whole-body tissue concentrations associated with adverse effects on biochemical or physiological endpoints were not used, nor were studies that evaluated the toxicity of chemical mixtures.

- **Tier 2** – For those COIs for which an insufficient number of studies was available – either fewer than 20 studies were

reviewed in Appendix B of the Ecological PRE or no fifth percentile screening was reported in Dyer et al. (2000) – the tissue residue TRV was calculated as the product of EPA AWQC and a bioconcentration factor (BCF).[2]

The Tier 2 approach is a simple bioaccumulation model that uses an AWQC concentration multiplied by a BCF to develop a TRV. AWQC (EPA 2002) were used, when available. Where current AWQC were not available, AWQC presented in the EPA Gold Book (the mid-1980s version of the national aquatic life criteria), as provided by EPA were used (EPA 2005). BCFs were based on aquatic species BCFs using K_{OW}s (octanol-water partitioning coefficients) developed according the methodology in Dyer et al. (2000) and Devillers (1996), as provided by EPA (2005).

Table 2-2 presents the whole-body critical tissue residue SL TRVs used for identifying fish tissue Round 2 COPCs based on the aquatic tissue TRVs used in the Ecological PRE. EPA's comments on the Ecological PRE state that "The aquatic TRVs presented in the Ecological PRE are acceptable for use in the Round 2 Comprehensive Report with the possible exception of chemicals for which the LWG did their own literature fifth percentile derivation" (EPA 2006c). EPA provided further direction (EPA 2006a) on how aquatic tissue TRVs for four aquatic tissue TRVs developed in the Ecological PRE based on LWG data (i.e., for total DDTs, cadmium, total PCBs, and 2,3,7,8-TCDD) should be derived for this ERA, and these comments were incorporated into the development of fish tissue SL TRVs, as noted in the Table 2-2.

Fish tissue SL TRVs were available for all fish COIs except for butyltin and benzyl alcohol. Tributyltin (TBT) is the most toxic butyltin, and the assessment of TBT is assumed to be protective of fish with regard to other butyltins (i.e., butyltin ion). In this ERA, no TBT tissue data were available for any fish receptors, with the exception of juvenile chinook salmon; therefore, the risk from TBT was evaluated for the critical tissue residue line of evidence for juvenile chinook salmon only.

Maximum tissue concentrations for each representative fish species were compared to the selected SL TRVs. COIs for which maximum tissue concentrations exceeded the selected SL TRV were identified as Round 2 COPCs. Results of the Round 2 COPC screen for each fish receptor are presented in the following subsections.

2.3.1 Largescale Sucker

Largescale sucker tissue composites were analyzed for all COIs except for butyltins, dioxins, furans, and dioxin-like PCB congeners (Table 2-3). The maximum concentration of seven COIs exceeded SL TRVs for largescale sucker: BEHP, dibutyl

[2] For 4-methylphenol, no AWQC was available, and the SL TRV was based on the lowest measured adverse effect tissue residue (LOAEL) presented in the Appendix B of the Ecological PRE (Windward 2005).

phthalate, total PCBs, 4,4′-DDD, beta-hexachlorocyclohexane (HCH), delta-HCH, and total DDTs. Three of these SL TRV exceedances (for dibutyl phthalate, beta-HCH, delta-HCH) were based on detection limits, and these COIs were not detected in largescale sucker whole-body tissue. These COIs are discussed in the uncertainty analysis presented in Section 4.0 of Appendix G, and were not retained as COPCs for largescale sucker. All other COIs with exceedances of SL TRVs (i.e., BEHP, total PCBs, 4,4′-DDD, and total DDTs) were retained as COPCs for largescale sucker.

2.3.2 Carp

Carp was evaluated as a surrogate ecological receptor for dioxins and dioxin-like chemicals only. The maximum concentrations of 2,3,7,8-TCDD was less than the SL TRV (Table 2-4). Therefore, no tissue Round 2 COPCs were retained for carp.

2.3.3 Sculpin

Sculpin tissue composites were analyzed for all COIs except butyltins. The maximum concentrations of nine COIs in sculpin tissue exceeded their respective SL TRVs: BEHP, dibutyl-phthalate, hexachlorobutadiene, total PCBs, 4,4′-DDD, 4,4′-DDT, beta-HCH, delta-HCH, and total DDTs (Table 2-5). Three of these SL TRV exceedances (for dibutyl-phthalate, delta-HCH, and hexachlorobutadiene) were based on detection limits; however, these COIs were either not detected in tissue or maximum detected tissue concentrations were below SL TRVs. Therefore, these COIs were not retained as Round 2 COPCs for sculpin and are discussed in the uncertainty analysis presented in Section 4.0 of Appendix G. The SL TRV exceedance for beta-HCH was based on detection limits; however, the maximum detected concentration also exceeded the SL TRV. This COI and the other five COIs with exceedances of SL TRVs (BEHP, total PCBs, 4,4′-DDT, 4,4′-DDD, and total DDTs) were retained as Round 2 COPCs for sculpin.

2.3.4 Peamouth

Peamouth was evaluated as an alternative receptor to largescale sucker (SEA et al. 2002), so peamouth tissue was analyzed for only a subset of the chemicals analyzed in largescale sucker tissue. Peamouth tissue composites were not analyzed for butyltins, dioxins, furans, PCB congeners, phthalates, phenols, and SVOCs. Only one of the COIs analyzed (lead) exceeded the SL TRVs for peamouth (Table 2-6). Lead was the only Round 2 COPC retained for peamouth.

2.3.5 Juvenile Chinook Salmon

Juvenile chinook salmon tissue composites were analyzed for all COIs. The maximum concentrations of five COIs exceeded their respective SL TRVs for juvenile chinook salmon: zinc, BEHP, butylbenzyl phthalate, dibutyl phthalate, and 4,4′-DDD (Table 2-7). Three of these SL TRV exceedances (for BEHP, butylbenzyl phthalate and dibutyl phthalate) were based on detection limits representing the

maximum concentration, however, detected tissue concentrations of these COIs did not exceed SL TRVs or these COIs were never detected in juvenile chinook salmon tissue. Therefore, these COIs are discussed in the uncertainty analysis presented in Section 4.0 of Appendix G, and were not retained as Round 2 COPCs for juvenile chinook salmon. The other two COIs that exceeded the SL TRVs (zinc and 4,4′-DDD) were retained as Round 2 COPCs for juvenile chinook salmon.

2.3.6 Smallmouth Bass

Smallmouth bass tissue composites were analyzed for all COIs except butyltins. The maximum concentrations of seven COIs exceeded their respective SL TRVs for smallmouth bass: BEHP, dibutyl phthalate, total PCBs, 4,4′-DDD, beta-HCH, delta-HCH, and total DDTs (Table 2-8). Three of these SL TRV exceedances (for beta-HCH, delta-HCH, and dibutyl phthalate) were based on detection limits representing maximum concentrations; however, these COIs were never detected in smallmouth bass tissue. These COIs are discussed in the uncertainty analysis presented in Section 4.0 of Appendix G, and were not retained as Round 2 COPCs for smallmouth bass. The other four COIs that exceeded the SL TRVs (BEHP, total PCBs, 4,4′-DDD, and total DDTs) were retained as Round 2 COPCs for smallmouth bass.

2.3.7 Northern Pikeminnow

Northern pikeminnow was evaluated as an alternative receptor to smallmouth bass (SEA et al. 2002), so northern pikeminnow tissue was analyzed for only a subset of the chemicals analyzed in smallmouth bass tissue. Northern pikeminnow tissue composites were not analyzed for butyltins, dioxins, furans, dioxin-like PCB congeners, phthalates, phenols, and SVOCs. The maximum concentrations of five COIs exceeded their respective SL TRVs for northern pikeminnow: mercury, total PCBs, beta-HCH, delta-HCH, and total DDTs (Table 2-9). Two of these SL TRV exceedances (for beta-HCH and delta-HCH) were based detection limits representing the maximum concentration; however, these COIs were never detected in northern pikeminnow tissue. Therefore, these COIs are discussed in the uncertainty analysis presented in Section 4.0 of Appendix G, and were not retained as Round 2 COPCs for northern pikeminnow. The other three COIs that exceeded the SL TRVs (mercury, total PCBs, and total DDTs) were retained as Round 2 COPCs for northern pikeminnow.

2.3.8 Summary of fish tissue Round 2 COPCs

A summary of the fish tissue Round 2 COPCs that were evaluated for each fish receptor is presented in Table 2-10. These Round 2 COPCs were evaluated in the fish risk assessment (see Section 4.0 of Appendix G).

The following COI/receptor pairs had non-detected concentrations that were greater than SL TRVs based on the maximum tissue concentration being represented by a detection limit; however, maximum detected concentrations of these COIs did not exceed SL TRVs or these COIs were not detected:

- BEHP/juvenile chinook salmon
- Butylbenzyl phthalate/juvenile chinook salmon
- Dibutyl phthalate/largescale sucker
- Dibutyl phthalate/sculpin
- Dibutyl phthalate/juvenile chinook salmon
- Dibutyl phthalate/smallmouth bass
- Hexachlorobutadiene/sculpin
- beta-HCH/largescale sucker
- beta-HCH/smallmouth bass
- beta-HCH/northern pikeminnow
- delta-HCH/largescale sucker
- delta-HCH/sculpin
- delta-HCH/smallmouth bass
- delta-HCH/northern pikeminnow

These COI/receptor pairs were not retained as COPC/receptor pairs because SL TRV exceedances are based on detection limits. Risk associated with detection limits that exceed TRVs are discussed in the uncertainty analysis presented in Section 4.0 of Appendix G.

3.0 PREDICTED TISSUE ASSESSMENT

An additional LOE in the measure of exposure and effects evaluating fish tissue data called for predicted tissue data to be compared to tissue-based TRVs in order to ensure that no COIs were missed due to the limited number of tissue samples analyzed.

The predicted tissue LOE was developed in accordance with the draft EPA guidance (Burkhard 2006). The co-located sediment and tissue data for field-collected sculpin were used to develop biota-sediment accumulation factors (BSAFs). Only chemicals with detected concentrations in both sediment and tissue samples were included[3]. Sediment and tissue concentrations of organic COIs were organic carbon and lipid normalized, respectively (Table 3-1). PAHs were not evaluated in sculpin tissue, because they are extensively metabolized and evaluated in other LOEs. Table 3-1 presents a summary of COIs for sculpin for the predictive tissue line of evidence. The co-located data were visually and statistically evaluated using scatter plots of tissue concentration versus sediment concentrations and BSAF versus sediment concentration. If the BSAF was found to be independent of sediment concentration, the average was determined of the BSAFs calculated for each co-located sediment and tissue concentration. If no relationship or an inverse relationship existed between the sediment and tissue concentration, no BSAF was developed. If the BSAF varied with sediment concentration, linear and log-linear regression models were fitted to the co-located data and the best fitting model[4] was selected. This approach was performed for all COIs except, if the BSAF decreased as the sediment concentration increased and the tissue concentrations at the higher sediment concentration were non-detects, in which case a BSAF was not determined.

COPCs were identified by multiplying the 95th percentile of the site-wide sediment concentration by the BSAF and compare the result to the SL TRV. If the predicted 95th percentile tissue concentration exceeded the SL TRV, the chemical was retained as a COPC.

Two chemicals, selenium and BEHP, were identified as Round 2 COPCs for sculpin based on the co-located field-collected sculpin tissue and sediment data (Table 3-1). The risks from these two Round 2 COPCs are further evaluated in Section 3.0 of Appendix G.

[3] PCBs, dioxins and furans, and DDTs were evaluated using the FWM (see Appendix E).
[4] The best fitted model was determined based on R^2 values, visual inspection of residuals, and distribution of the independent variable.

4.0 DIETARY DOSE ASSESSMENT

The dietary dose assessment was another line of evidence for evaluating risks from metabolized and regulated chemicals to selected fish receptors. Per EPA comments to the Ecological PRE (EPA 2006c), the dietary assessment for fish was based on a dietary dose approach, rather than on estimated dietary concentrations. In the dietary assessment, daily dietary doses were estimated for each fish receptor and dietary chemical pair. Dietary dose estimates included ingestion of biota (prey) and incidental ingestion of sediment. Dietary doses were estimated using the following equation:

$$IR_{diet} = \frac{(FIR \times C_{prey}) + (SIR \times C_{sed})}{BW} \times SUF \qquad \textbf{Equation 4-1}$$

Where:

IR_{diet}	=	estimated fish ingestion rate or exposure dose (mg/kg bw/day)
FIR	=	food ingestion rate (kg ww food/day)
C_{prey}	=	tissue concentration in prey items (mg/kg ww)
SIR	=	sediment ingestion rate (kg dw sediment/day)
C_{sed}	=	concentration in surface sediment (mg/kg dw)
BW	=	body weight (kg)
SUF	=	site use factor (unitless); fraction of time that a receptor spends foraging at the Study Area relative to their entire home range

Body weights, food ingestion rates, sediment ingestion rates, and site use factors vary among fish receptors.

This section presents the initial screening process for the dietary dose assessment, including the dietary screen and Round 2 COPC identification. These Round 2 COPCs were evaluated in the fish risk assessment (see Section 4.0 of Appendix G).

4.1 DIETARY SCREEN

To identify Round 2 COPCs for fish, the maximum estimated exposure dose (IR_{diet}) of each dietary fish COI was compared to a NOAEL TRV for that chemical. If the maximum exposure dose (for each receptor) was greater than the NOAEL TRV, the chemical was identified as a COPC/receptor pair for this Round 2 ERA. The identification of Round 2 COPCs through a conservative screen in which maximum concentrations in relevant media are used is consistent with EPA guidance (EPA 1997a, b) and an important step for narrowing the list of chemicals to only those that could potentially pose a risk to ecological receptors.

4.2 SELECTION OF COIS

Fish dietary COIs were defined as a subset of those fish tissue COIs (Table 2-1) that are metabolized or regulated in fish. These dietary fish COIs include metals, PAHs,

and butyltins. In addition, total PCBs, mercury, and total DDTs were evaluated as dietary fish COIs, as agreed upon by LWG and EPA.

4.3 IDENTIFICATION OF ROUND 2 COPCS

Fish dietary dose NOAEL TRVs were developed for all fish dietary COIs when literature was available. NOAEL TRVs were based on an extensive search of the available toxicological literature as presented in the Ecological PRE (Windward 2005). For this Round 2 ERA, the assumptions used in the dose-based TRVs (specifically, fish ingestion rates) presented in the Ecological PRE were revisited. For those studies in which neither a dietary dose nor a feeding rate were presented, the dose was estimated using the average feeding rate from toxicological studies for the same fish species fed a similar food type (i.e., live prey or a synthetic diet). Dietary dose NOAEL TRVs were selected based on the updated dietary dose TRV calculations to represent conservative screening level toxicological thresholds. The TRV technical memorandum addendum (Attachment G5) presents the updated dietary dose TRVs, including a detailed discussion of how TRVs were derived and the identification of selected dietary NOAEL TRVs.

Dietary dose NOAEL TRVs were developed for 12 fish COIs based on available toxicity data. No dietary NOAEL TRVs were identified for four COIs: antimony, chromium, nickel, and thallium (Table 4-1).

The majority of the selected NOAEL TRVs those are based on the NOAEL TRVs recommended in the TRV technical memorandum addendum (Attachment G5). For COIs for which no chronic NOAEL TRVs was selected or the selected NOAEL was not based on a chronic study, uncertainty factors (UFs) were used to derive chronic NOAELs. EPA and its partners recommended using UFs to develop conservative TRVs in a data screening evaluation (EPA 2006c). UFs for the Round 2 ERA were based on EPA Region 10 guidance and used to extrapolate a TRV where a chronic NOAEL was not selected (Table 4-2). UFs, also called safety factors, are often used to provide conservative screening levels and/or risk estimates when more specific toxicological information is lacking. They are used to extrapolate toxicity data to a desired effect level, exposure duration, and species when no data are available (Duke and Taggart 2000). NOAEL TRVs are presented in Table 4-3.

To identify dietary dose Round 2 COPCs for fish, NOAEL TRVs were compared to the estimated maximum dietary dose (IR_{diet}) for each of the seven fish receptors. The maximum dietary dose was calculated as milligrams of each Round 2 COPC ingested per kilogram of body weight per day (mg/kg bw/day) using Equation 4-1. The maximum daily ingested doses for largescale sucker, white sturgeon, sculpin, peamouth, juvenile chinook salmon, smallmouth bass, and northern pikeminnow were calculated using the maximum concentrations in any of the associated prey species in the ERA dataset. Table 4-4 presents the prey identified for each receptor. The

maximum[5] surface sediment concentration (representing C_{sed}) and the maximum tissue concentration (representing C_{prey}) in any of the prey species identified for each receptor were used to estimate maximum dose (IR_{diet}). The exposure parameters (body weight, food ingestion rate, and sediment ingestion rate) and dietary prey items used to calculate the maximum dietary exposure doses for each fish receptor are presented in Table 4-5 and described in detail in Section 4.0 of Appendix G. An SUF of 1.0 was assumed for all fish receptors for the identification of Round 2 COPCs.

Estimates of the maximum daily ingested doses of five COIs (i.e., cadmium, copper, TBT, total PCBs, and total DDTs) were greater than their respective NOAEL TRVs for all fish receptors (Tables 4-6 through 4-12). The maximum daily dose of mercury was greater than the fish dietary dose NOAEL TRV for northern pikeminnow. These Round 2 COPCs/receptor pairs were evaluated in Section 4.0 of Appendix G. In addition, maximum daily ingested doses of total PAHs were greater than the NOAEL TRV for largescale sucker and pre-breeding sturgeon; and the maximum daily ingested doses of arsenic, mercury, and selenium were greater than their respective NOAEL TRVs for largescale sucker. These Round 2 COPCs/receptor pairs were evaluated in Section 4.0 of Appendix G. A summary of the dietary Round 2 COPCs that were evaluated for each fish receptor are presented in Table 4-13.

[5] The maximum sediment and tissue concentrations were represented by the maximum detected concentration or the maximum RL, whichever was higher.

5.0 WATER ASSESSMENT

In the water assessment, the exposure of fish was evaluated by comparing detected chemical concentrations in surface water and TZW collected at the Study Area to chemical concentrations in water known to be toxic to fish. The results of the surface water screen are presented in Section 4.1. The results of the transition zone water screen are presented in Section 4.2. Per EPA request, an additional line of evidence was evaluated where metal concentrations in surface water were compared to concentrations in water associated with olfactory function impairment in migrating adult chinook salmon populations. The results of this line of evidence are presented in Section 4.3.

5.1 SURFACE WATER SCREEN

This section presents the initial screening process for surface water assessment including screening of surface water data and Round 2 COPC identification.

5.1.1Selection of COIs

COIs for fish were defined as all surface water chemicals detected in the Study Area, excluding crustal elements (i.e., aluminum; see Section 2.0 of Appendix G). The list of surface water COIs for fish is presented in Table 5-1.

5.1.23 Identification of Round 2 COPCs

To identify Round 2 COPCs for fish, maximum detected COI concentrations in surface water were compared to the ecological surface water screening level (Eco SL) for that chemical. If the maximum concentration was greater than the Eco SL, the chemical was identified as a Round 2 COPC.

A review of water quality regulatory benchmarks and literature-based thresholds was conducted to develop the Eco SLs for chemicals in water. Eco SLs were developed for all fish surface water COIs except individual dioxins and furans other than 2,3,7,8-TCDD because no data were available. Eco SLs, presented in Attachment G3, were developed by LWG and revised based on EPA's comments (EPA 2006b). LWG adopted EPA's proposed hierarchy for Eco SL selection (LWG 2006) and in accordance with EPA's comments, a UF of 50 was used to calculate a chronic screening value from an acute screening value when no chronic data were available. There is high uncertainty in the use of a UF of 50, and LWG recommends the use of a range of UFs for the BERA. The Eco SLs are considered to be protective of all aquatic receptors, including benthic invertebrates, fish, and amphibians.

The Round 2 COPC screening results are presented in Table 5-2. Based on the results of the surface water screening, the following 10 COIs had maximum surface water concentrations greater than chronic Eco SLs and were retained as surface water Round 2 COPCs for fish:

- Zinc (dissolved)
- Benzo(a)anthracene
- Benzo(a)pyrene
- 4-Chloro-3-methylphenol
- Total PCBs
- 2,4′-DDD
- 2,4′-DDT
- 4,4′-DDD
- 4,4′-DDT
- Total DDTs

5.2 TRANSITION ZONE WATER SCREEN

The identification of TZW Round 2 COPCs for fish receptors is the same as that for benthic invertebrates (Section 3.0), in which exposure was evaluated by comparing maximum detected chemical concentrations in TZW to Eco SLs. The list of TZW Round 2 COPCs is repeated in Table 5-3.These chemicals were retained as TZW Round 2 COPCs for fish.

5.3 CHINOOK OLFACTORY FUNCTION ASSESSMENT

Per EPA request (cite LOE table), an additional line of evidence was evaluated for surface water pathway. Specifically, the exposure of adult chinook salmon to metal concentrations in water was evaluated to determine if these concentrations might disrupt olfactory function in migrating populations. Adult chinook salmon are not a selected ecological receptor of concern; however, this receptor was evaluated per EPA request. Impaired olfactory function and avoidance response behaviors have been associated with concentrations of metals in water. Avoidance toxicity studies of salmonids exposed to metals in water was limited to two chemicals (i.e., copper and cobalt), but cobalt was not analyzed in Portland Harbor surface water. Therefore, only copper was evaluated.

5.3.1 Exposure of Adult Chinook Salmon

Surface water concentrations of dissolved copper at the Study Area ranged from 0.37 to 1.64 µg/L. Adult chinook salmon (from both spring and fall runs) are exposed to these copper concentrations as they return from the ocean and migrate through the Study Area, making their way upstream to spawn. The migration of adult spring

chinook salmon through the Multnomah Channel and the Willamette mainstem downstream from Willamette Falls peaks in mid- to late April and is mostly complete by July (ODFW 2001; Schreck et al. 1994). In 2001, 85 to 95% of spring chinook salmon returning to the Willamette Basin were raised in hatcheries (ODFW 2001), although wild populations of spring chinook salmon also migrate through the Study Area to spawn upstream, above Willamette Falls and in the Clackamas River. The migration speed and distance of adult spring chinook is variable and may be related to the time of migration (Schreck et al. 1994).

5.3.2 Impaired Olfactory Function in Salmonids

Chinook salmon olfactory function can be affected by exposure to dissolved metals and other contaminants in water (Klaprat et al. 1992; Baldwin et al. 2003). Olfactory function in fish plays a major role in mediating behaviors important for both survival and reproduction, such as juvenile imprinting on home waters, predator avoidance, and adult migration and homing (Baldwin et al. 2003). Typically, the presence of metals in water produces avoidance behavior in fish (Hansen et al. 1999b; Atchison et al. 1987). Field studies have shown that in cases where distinct spatial gradients of metals occur (e.g., near point-source discharges), fish may use their sense of smell to avoid contamination (Saunders and Sprague 1967). Lorz and McPherson (1976) reported that the exposure of 18-month-old coho salmon to 5 µg/L of copper resulted in a 30% reduction in downstream migration. Similarly, Baldwin et al. (2003) reported that the presence of metals in water may disrupt migration patterns or prevent fish from inhabiting areas that would otherwise offer productive habitat. Impairment of the olfactory system is temporary if the exposure lasts for no more than a few moments. However, the death of olfactory cells has been shown to occur following exposures lasting more than 4 hours (Hansen et al. 1999a; Julliard et al. 1996). Although it is possible for fish to regenerate olfactory cells, this re-growth takes several weeks, during which time the fish is not receiving important information usually conferred by those cells.

The exposure of salmonids to individual metals in laboratory studies at levels of contamination below those known to be lethal has been shown to produce avoidance behavior. A literature review was conducted of laboratory studies involving salmonids and exposure to metals in water in which avoidance behavior was observed, and the results are summarized in Table 5-4. Based on the reviewed literature, salmonids displayed avoidance behavior at copper concentrations ranging over several orders of magnitude, from 0.10 to 88 µg/L. Hardness affects copper chemistry; therefore, hardness-effects studies within a hardness range similar to that of the LWR (i.e., 22.5 to 54.5 mg/L) are more pertinent for affects on avoidance behavior.

5.3.3 Relevance of Avoidance Response on Migration

Salmonid populations can survive in environments with dissolved copper concentrations that range from 2 to 23 µg/L (e.g., Copper River, Alaska; Sacramento

River, California) (Brooks 2004). Adult and juvenile salmonids survive and successfully navigate in both of these complex river systems, which frequently have dissolved copper concentrations greater than 2 µg/L, suggesting that olfactory inhibition may be minimal for salmonid populations that are acclimated to elevated copper levels. Other studies have suggested, alternatively, that acclimation to dissolved copper concentrations above 2 µg/L causes the loss of avoidance behavior at higher copper concentrations (Brooks 1998; Hansen et al. 1999b) (Brooks, 2000), with the result that salmonids may not avoid lethal concentrations.

5.3.4 Evidence of Olfactory-Associated Migration Effects

Adult chinook salmon olfactory-associated effects were evaluated, per EPA request, by comparing copper concentrations associated with salmonid avoidance response behavior to copper concentrations measured in surface water at the Study Area. Two endpoints are associated with the avoidance response: impaired reproduction (due to olfactory avoidance creating barriers to spawning) and direct toxicity (due to loss of olfactory sensitivity, and therefore lack of avoidance of elevated copper concentrations). The direct toxicity endpoint is already addressed in the Round 2 report because no copper avoidance is assumed, so it is not addressed further here.

Concentrations of dissolved copper at the Study Area ranged from 0.37 to 1.64 µg/L (need to qualify). This site-wide range of copper concentrations are at the low end of the range of copper concentrations associated with salmonid avoidance response in laboratory studies, 0.10 to 88 µg/L. Field studies have shown mixed results on whether copper concentrations > 2 µg/L result in impaired olfactory function in fish. Therefore, it is highly unlikely that copper concentrations at the Study Area would cause a change in migratory behavior. Given the site-specific observations of both juvenile and adult salmon passage through the LWR, the actual empirical evidence demonstrates there is an unlikely adverse effect from copper concentrations in surface water at the Study Area. Furthermore, these copper concentrations are typical of regional levels; based on summary statistics from the USGS National Stream Quality Accounting Network (NASQAN) program, water copper concentrations at six sites along the Columbia River Basin from 1996 to 2003 ranged from <1.0 µg/L to 6.7 µg/L (USGS 2006). Therefore, the pathway associated with reproductive impairment is incomplete and the potential risk from olfactory copper avoidance is insignificant. This line of evidence was not further evaluated in the risk assessment for fish receptors.

6.0 ASSESSMENT OF THE OCCURRENCE OF LESIONS AND PAHS IN SEDIMENT

Per EPA request (EPA 2006a) (see Section 4.1.4 of Appendix G), an additional line of evidence was evaluated for the exposure of benthic fish to PAHs. This assessment involved comparing PAH sediment concentrations at the Study Area to literature-derived PAH concentrations associated with the occurrence of skin or liver lesions.

Fish are most likely exposed to PAHs through direct contact with bottom sediments and sediments that are suspended in water and through dietary uptake (Johnson et al. 2002). Fish exposed to PAH-contaminated sediments through direct contact have been shown to have increased incidence of skin and liver lesions as well as other deformities (Myers et al. 1994; Pinkney et al. 2000). In addition, reduced lifespan in fish has been linked to cancerous lesions (Johnson et al. 2002; Baumann et al. 1987; Pinkney et al. 2000; Myers et al. 1994). The prevalence of both hepatic and epidermal lesions can be used as a criterion for identifying contaminated sites (Pinkney et al. 2004a).

Section 5.1 presents PAH sediment concentrations in the Study Area and field observations of fish collected from within the Study Area. Section 5.2 presents a review of the toxicological literature. These data were then compared in Section 5.3.

6.1 STUDY AREA-SPECIFIC DATA

Total PAH[6] concentrations in surface sediment at the Study Area ranged from 5.0 to 7,258,000 µg/kg. The surface weighted average concentration (SWAC) of total PAH in sediment is 24,285 µg/kg. Benthic fish are exposed to PAH-contaminated sediment while foraging. Field notes collected during Round 1 sampling included the following observations of fish abnormalities: fin hemorrhage and erosion, frayed fins, red and purple lumps, reddening of the anal area, reddening and paling of the maxillary hinges, and external tears and lacerations in various locations on the specimens (SEA et al. 2003).

As part of the US Geological Survey (USGS) Biomonitoring of Environmental Status and Trends (BEST) program, fish were collected from September 1997 to April 1998 from 16 stations in the Columbia River Basin. During fish sampling, observations of external and internal lesions were made and data were collected on health and reproductive biomarkers (Hinck et al. 2004). The fish (i.e., common carp, bass, largescale sucker, northern pikeminnow, and walleye) were collected from two locations in the Willamette River, at RM 10 (within the Study Area) and at RM 30 (above the Study Area). Of all the fish collected in the Columbia River Basin study,

[6] Total PAHs are the sum of concentrations for the following chemicals: acenaphthylene, acenaphthene, anthracene, benzo(a)anthracene, total benzofluoranthenes, benzo(a)pyrene, benzo(g,h,i)perylene, chrysene, dibenzo(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3,-c,d)pyrene, naphthalene, phenanthrene, and pyrene.

74% were found to have external gross lesions (abnormalities), which were categorized by location (i.e., on body surface, eyes, opercles, or fins). Eighty-eight percent of fish collected at RM 10 and eighty-five percent of fish collected at RM 30 were found to have external gross lesions. Most fish lesions were located on fish fins. As noted in the study, "The percent of lesions for these fish are high and caution needs to be used when interpreting these data. Many lesions identified are not a result of exposure to environmental contaminants but are more likely due to holding time in nets prior to fish processing or normal wear as a fish ages" (Hinck et al. 2004). Thus, the observation of lesions in Round 1 and USGS fish collected from the Study Area are inconclusive and may likely be the result of fish collection and handling rather than an indicator of fish health.

6.2 TOXICOLOGICAL LITERATURE REVIEW

This section discusses the toxicological studies that examined PAH exposure and an increase in the incidence of lesions. Section 4.6.2.1 summarizes studies that involved external lesions and other deformities; Section 4.6.2.2 summarizes studies that reported on hepatic lesions. The implications of lesion prevalence for population-level effects are discussed in Section 4.6.2.3.

6.2.1 Toxicological Studies on External Lesions and Other Deformities

External lesions observed in fish exposed to PAHs in sediment included epidermal papillomas, mucoid plaques, and lip papillomas (Baumann et al. 1996), as well as epidermal ulcerations/abrasions and fin lesions (Mezin and Hale 2000). In Mezin and Hale, one group of mummichogs was exposed to PAH-contaminated sediment from the Elizabeth River Superfund site, and another group was exposed to uncontaminated sediment for 13 days. Ninety-four percent of the lesions observed on test fish were in the PAH-exposed group. Of these,74% of the lesions were fin erosions; others manifested as epidermal ulcerations and abrasions, primarily in the anal region (Mezin and Hale 2000). In some cases, lesions were so severe that internal organs were exposed. Although specific sediment PAH concentrations were not reported in this study, PAH concentrations in Elizabeth River sediments have been reported at levels up to 170,000 µg/kg (Vogelbein and Unger 2003).

In a toxicological field study, Pinkney et al. (2004a) surveyed brown bullhead from the Anacostia River, which had total PAH concentrations in sediment that ranged from 15.2 to 30.9 µg/g.[7] Tumor prevalence in Anacostia fish were compared with tumor prevalence in fish from an uncontaminated control site with an average total PAH concentration in sediment of 190 µg/kg. Skin lesion prevalence in fish from the Anacostia site ranged from 13 to 23% in large brown bullhead versus 0% in fish from the reference site. Pinkney et al. (2004a; 2004b) linked the prevalence of skin tumors to PAH biomarkers in bullhead from three PAH-contaminated Chesapeake Bay

[7] Sediment PCBs and chlordane concentrations, which were also elevated within the Study Area, were not reported.

rivers. A significantly higher prevalence of skin tumors was observed in fish from the contaminated river sites than in the fish from the reference site. The average total PAH concentration in sediment at the reference site was 187 µg/kg, while total PAH concentrations in sediment from the Chesapeake Bay rivers were elevated to within the range of 6,480 to 6,750 µg/kg.

6.2.2 Toxicological Studies on the Prevalence of Hepatic Lesions

The link between PAH exposure and hepatic lesions is well-documented. A statistical study of eight investigations that explored the link between PAH contaminants in sediment and hepatic and kidney lesions in fish reported that overall hepatic lesion prevalence ranged from 4 to 16%, with the highest prevalence occurring at Eagle Harbor, Washington, in the Puget Sound (Landahl et al. 1990). The PAH concentration in sediment at this location was 540,000 µg/kg dw, higher than at any other location in the other studies. When six of the eight studies were considered as a whole, Landahl et al. found consistent relationships between PAH sediment concentrations and the development of hepatic lesions in four of the five hepatic lesion categories considered in the study.

When PAHs are metabolized in the livers of fish, metabolites are produced. Some of these metabolites are carcinogenic, mutagenic, or cytotoxic and are thought to be linked to hepatic lesion development (Malins et al. 2006); citing Maccubbin 1994-**waiting to have entered into EndNote**);(Johnson 2000; as cited in Malins et al. 2006; Myers et al. 1994; Pinkney et al. 2004a). Pinkney et al. (2000) reported that the probability of brown bullhead developing liver hepatocarcinoma increased with an increase in bile PAH metabolite concentrations by a factor of 1.0 with each 100,000 µg/kg metabolite concentration increase.

Several studies that established a relationship between elevated sediment PAH concentrations and liver lesions were used to conduct a hockey stick regression to determine sediment PAH effects concentration thresholds for several lesion types (Johnson et al. 2002). Lesion classes included neoplasms and a category of "any lesions." The total PAH sediment concentration threshold for effects (and confidence limits) was calculated as 2,800 µg/kg (11 to 5,500 µg/kg)[8] for neoplasms and 620 µg/kg (300 to 1,000 µg/kg) for the "any lesions" category (Johnson et al. 2002).

Stern et al. (2003) conducted a separate hockey stick regression analysis of PAH concentrations in Puget Sound sediment using the "any lesion" category as defined in Johnson et al. (2002) and lesion data from that study as well as other lesion data. In this analysis, concentrations in sediment were characterized as the total PAH SWAC [9] over an assumed English sole foraging radius of 1 km. The resulting total PAH sediment effects concentration threshold for the "any lesion" category (and confidence limits) was 2,731 µg/kg (1,410 to 3,772 µg/kg).

[8] Total PAHs were the sum of 10 low molecular weight PAHs and 8 high molecular weight PAHs.
[9] Based on the sum of 10 high molecular weight PAHs.

6.2.3 Implications of Lesion Prevalence for Population-Level Effects

Although several studies have linked sediment PAH exposure to lesion prevalence, few have linked lesion prevalence to adverse effects at the population level. Several brown bullhead studies conducted on the Black River in Ohio reported liver histopathology data that shows a link between sediment PAH concentrations, liver lesions, and population age structure (Baumann 2000). In the early 1980s, PAH concentrations in Black River sediments were as high as several hundred parts per million (Baumann 2000, citing Baumann et al. 1982), largely the result of discharge from a steel and coke plant. Under these conditions, brown bullhead in the study area showed a high prevalence of liver lesions, with less than 20% having completely normal livers. A truncated age structure, whereby few individuals in the population survived beyond 4 years of age, was also observed. Baumann et al. (1987) reported that liver tumor prevalence increased significantly with fish age and attributed the truncated age structure to a reduced lifespan resulting from cancerous lesions.

Surveys conducted subsequent to the plant's closure in 1982 showed that tumor frequency in fish aged 3 and older declined by approximately 50%, and the incidence of cancer was reduced to approximately 25% of earlier levels (Baumann 2000, citing Baumann and Harshbarger 1995). In addition, the age structure of the fish population shifted, with more aged 5 fish captured and aged 6 fish showing up in the surveys for the first time. After remedial dredging and a recovery period, sediment sampling in the Black River in 1997 and 1998 showed that nearly all PAH concentrations in sediment were down to 15,000 µg/kg or less. PAH concentrations in fish tissue were also significantly lower than earlier levels. In 1998, cancer prevalence in fish aged 5 and older was 7%, and the age structure of the population more closely resembled that of populations in pristine sites, with over 60% of the fish population surveyed aged 5 or older and over 35% aged 6 or older (Baumann 2000). This analysis suggests that the population-level effects threshold for total PAHs may be greater than 15,000 µg/kg.

In a study of English sole, Johnson and Landahl (1994) examined the relationship between lesion prevalence and population-level effects in a study of estimated annual mortality rates at both highly contaminated (e.g., Eagle Harbor) and uncontaminated sites throughout Puget Sound. English sole mortality rates from contaminated sites associated with high liver lesion prevalence were not found to be significantly greater than mortality rates for English sole from Puget Sound as a whole. Johnson and Landahl also examined the English sole population structure and found no evidence of increased age-related mortality in fish with lesions or in populations associated with areas of high levels of PAHs and PCBs. The authors concluded that fish populations that have high incidence of lesions do not necessarily have increased mortality. Other factors that effect English sole populations, such as fishing pressure, predation, and fluctuations in food supply may mask population-level effects associated with chemical contamination and lesion incidence. Thus, this study (Johnson and Landahl 1994) did not identify a link between lesion prevalence and

population structure in areas with widely varying ranges of PAH concentrations in sediment.

6.2.4 Other Factors Contributing to Lesion Prevalence

Several factors other than sediment PAH concentrations (e.g., age, sex, and other non-chemical stressors) have been shown to be significant risk factors for lesion prevalence in benthic fish (Baumann et al. 1987; Myers et al. 1994; Pinkney et al. 2000, 2004a). Age has been identified as an important risk factor for lesion development in brown bullhead, with the incidence of skin tumors increasing 2.5 times per year and the incidence of liver hepatocarcinoma increasing 3.5 times per year as fish age (Pinkney et al. 2000). Additional studies support the finding that age plays a role in lesion development in benthic fish (Baumann et al. 1987; Myers et al. 1994). The odds of developing liver hepatocarcinoma in brown bullhead has also been linked to sex, with a 4.5 times higher incidence of these lesions observed in female fish over male fish (Pinkney et al. 2000). The accumulation of bile PAH metabolites has been identified as a risk factor for lesions as well, with a 1.1 to 1.8 times increase in tumor prevalence observed per 100 mg/kg metabolite concentration increase (Pinkney et al. 2000). Additional factors including viruses, crowding, temperature change, and other biotic and abiotic factors may contribute to epidermal lesion development as environmental stressors suppress fish immune systems (Baumann et al. 1996).

Contaminants other than PAHs have been identified as risk factors for lesion development in benthic fish (Myers et al. 1994). PCB and DDT concentrations in sediment and liver tissues were found to be significant risk factors for neoplasms and pre-neoplastic lesions by Myers et al. (1994) and others (Stern et al. 2003; citing O'Niell et al. 1999).The role of PCBs in initiating neoplasms in fish is not well understood, and it is not clear if PCBs alone can induce lesions in wild fish. Although Myers et al. (1994) reported PAH exposure as the most frequently identified risk factor, PCBs, DDTs, and other contaminants were thought to be toxicologically relevant risk factors in the etiology of hepatic lesions. Because PAHs, PCBs, DDTs, and other contaminants typically coexist in contaminated sediments, it is difficult to quantify their relative contributions to the development of liver lesions in benthic fish (Myers et al. 1994).

6.2.5 Summary of Toxicological Studies

Several studies have proposed hepatic lesion sediment effects thresholds for PAH ranging from 230 to 4,000 µg/kg and spanning over an order of magnitude (Horness et al. 1998; Johnson et al. 2002; Stern et al. 2003). Johnson et al. suggested a 1,000 µg/kg PAH threshold as being be both protective of the majority of fish species and practical for making management decisions based on the “any lesions” category. Johnson et al. also reported a threshold of 2,800 µg/kg for neoplasms (benign or cancerous). Stern et al. (2003) reported an effects threshold of 2,700 µg/kg for the “any lesions” category and stated that a hockey stick regression using methods that

account for the size of English sole's foraging range would likely result in an even higher threshold for neoplasms. Pre- and post-dredge data from Eagle Harbor indicate that English sole hepatic lesion incidence was approximately equal to background levels at a SWAC in sediment of 4,000 µg/kg (Stern et al. 2003). Pre- and post-dredge data from the Black River indicate that brown bullhead lesion incidence and population structure were approximately equal to background levels at an area-weighted average sediment concentration of 15,000 µg/kg (Baumann 2000).

6.3 DIRECT CONTACT WITH PAHS IN SEDIMENT ASSESSMENT CONCLUSIONS

Direct contact of benthic fish to PAH concentrations was evaluated by comparing PAHs concentrations associated with lesion occurrence in field studies to total PAH concentrations measured in surface sediment at the Study Area. The spatially weight average of total PAH[10] concentrations at the Study Area is 24,285 µg/kg. This concentration is above effects thresholds reported by Johnson et al. (2002) and Stern et al. (2003), ranging from 230 to 4,000 µg/kg. However, these toxicological threshold concentrations reported in the literature have not been linked to population level effects.

The link between lesions and effects on fish populations have been demonstrated for brown bullhead from the Black River. However, the effects of specific classes of lesions (either cancerous or benign) and potentially pre-neoplastic lesions at the population level is uncertain and the appropriate estimation method for sediment thresholds for this or other endpoints are still under refinement. Total PAH concentrations in Study Area sediments are greater than post-remediation concentrations reported in the Black River (15,000 µg/kg) that have been associated with recovered population structure in brown bullhead; however, because pre-remediation concentrations in the Black River greatly exceeded those in Portland Harbor, it is not known if population effects may occur in the Study Area.

There is high uncertainty associated with literature-based PAH thresholds for fish lesion incidence due to: 1) the uncertainties associated with the determination of the appropriate spatial determination of PAH sediment exposure for specific benthic fish (e.g., whether sediment concentrations should be averaged across the area where fish are assumed to forage and be exposed to PAHs), 2) the potential confounding effects of co-occurring contaminants (Myers et al. 1994), and 3) because the link between population level effects of survival, growth, or reproduction and the occurrence of lesions in fish, and effect-level PAH concentrations in sediments has not been demonstrated in field studies. Due to the high uncertainty of this line of evidence,

[10] Total PAHs are the sum of the following chemicals: acenaphthylene, acenaphthene, anthracene, benzo(a)anthracene, total benzofluoranthenes, benzo(a)pyrene, benzo(g,h,i)perylene, chrysene, dibenzo(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3,-c,d)pyrene, naphthalene, phenanthrene, and pyrene.

PAH exposure to fish in sediment was not further evaluated in the risk assessment for fish receptors.

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TABLES

Table 2-1. Fish Tissue COIs

Chemicals of Interest	
Metals	
Antimony	Mercury
Arsenic	Nickel
Cadmium	Selenium
Chromium	Silver
Copper	Thallium
Lead	Zinc
Butyltins	
Butyltin ion	Tributyltin ion
PAHs	
2-Methylnaphthalene	Benzo(k)fluoranthene
Acenaphthene	Chrysene
Acenaphthylene	Dibenzo(a,h)anthracene
Anthracene	Fluoranthene
Benzo(a)anthracene	Fluorene
Benzo(a)pyrene	Indeno(1,2,3-cd)pyrene
Benzo(b)fluoranthene	Naphthalene
Benzo(b+j)fluoranthene	Phenanthrene
Benzo(g,h,i)perylene	Pyrene
Phthalates	
BEHP	Dibutyl phthalate
Butylbenzyl phthalate	Di-n-octyl phthalate
SVOCs	
Benzyl alcohol	Hexachlorobutadiene
Dibenzofuran	Hexachloroethane
Hexachlorobenzene	
Phenols	
4-Methylphenol	Phenol
Pentachlorophenol	
PCBs	
Total PCBs	

Table 2-1. Fish Tissue COIs

Chemicals of Interest	
Dioxins and Furans	
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin	1,2,3,7,8,9-Hexachlorodibenzo-*p*-dioxin
2,3,7,8-Tetrachlorodibenzofuran	1,2,3,7,8,9-Hexachlorodibenzofuran
1,2,3,7,8-Pentachlorodibenzo-*p*-dioxin	2,3,4,6,7,8-Hexachlorodibenzofuran
1,2,3,7,8-Pentachlorodibenzofuran	1,2,3,4,6,7,8-Heptachlorodibenzo-*p*-dioxin
2,3,4,7,8-Pentachlorodibenzofuran	1,2,3,4,6,7,8-Heptachlorodibenzofuran
1,2,3,4,7,8-Hexachlorodibenzo-*p*-dioxin	1,2,3,4,7,8,9-Heptachlorodibenzofuran
1,2,3,4,7,8-Hexachlorodibenzofuran	Octachlorodibenzofuran
1,2,3,6,7,8-Hexachlorodibenzo-*p*-dioxin	Octachlorodibenzo-*p*-dioxin
1,2,3,6,7,8-Hexachlorodibenzofuran	
Pesticides	
Aldrin	beta-Endosulfan
cis-Chlordane	Total endosulfan
Total chlordane	Endosulfan sulfate
2,4′-DDD	Heptachlor
2,4′-DDE	Heptachlor epoxide
2,4′-DDT	beta-HCH
4,4′-DDD	gamma-HCH
4,4′-DDE	delta-HCHe
4,4′-DDT	Methoxychlor
Total DDTs	cis-Nonachlor
trans-Chlordane	trans-Nonachlor
Dieldrin	Oxychlordane
alpha-Endosulfan	

BEHP – bis(2-ethylhexyl) phthalate
COI – chemical of interest
HCH – hexachlorocyclohexane
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound

Table 2-2. Fish Tissue SL TRVs Used to Determine Round 2 COPCs

COI	Unit (ww)	SL TRV	Source
Metals			
Antimony	mg/kg	0.03	estimated from AWQC and BCF
Arsenic	mg/kg	1.7	Dyer et al. (2000)
Cadmium	mg/kg	0.09[a]	fifth percentile LOAEL as presented in Appendix B of the Ecological PRE (Windward 2005)
Chromium	mg/kg	2.7[b]	estimated from AWQC and BCF
Copper	mg/kg	3.1	Dyer et al. (2000)
Lead	mg/kg	2.2	Dyer et al. (2000)
Mercury	mg/kg	0.46	Dyer et al. (2000)
Nickel	mg/kg	18.4	Dyer et al. (2000)
Selenium	mg/kg	1.1	Dyer et al. (2000)
Silver	mg/kg	0.27	Dyer et al. (2000)
Thallium	mg/kg	4.6	estimated from AWQC and BCF
Zinc	mg/kg	27	Dyer et al. (2000)
Butyltins			
Tributyltin ion (TBT)	µg/kg	49.9	estimated from AWQC and BCF
Phthalates			
BEHP	µg/kg	390	estimated from AWQC and BCF[c]
Butylbenzyl phthalate	µg/kg	1,200	estimated from AWQC and BCF
Dibutyl phthalate	µg/kg	270	estimated from AWQC and BCF
Di-n-octyl phthalate	µg/kg	41,000	estimated from AWQC and BCF
SVOCs			
Dibenzofuran	µg/kg	1,000	estimated from AWQC and BCF
Hexachlorobenzene	µg/kg	490	Dyer et al. (2000)
Hexachlorobutadiene	µg/kg	26	estimated from AWQC and BCF
Hexachloroethane	µg/kg	47,000	estimated from AWQC and BCF
Phenols			
4-Methylphenol	µg/kg	76,500	LC50 as presented in Appendix B of the Ecological PRE (Windward 2005)
Pentachlorophenol	µg/kg	3,100	estimated from AWQC and BCF
Phenol	µg/kg	6,200	estimated from AWQC and BCF
PCBs			
Total PCBs	µg/kg	720[c]	fifth percentile LOAEL as presented in Appendix B of the Ecological PRE (Windward 2005)
Dioxins and Furans			
2,3,7,8-TCDD	pg/g	90[d]	fifth percentile LOAEL as presented in Appendix B of the Ecological PRE (Windward 2005)
Pesticides			
Aldrin	µg/kg	810	estimated from AWQC and BCF
cis-Chlordane	µg/kg	550[g]	Dyer et al. (2000)
trans-Chlordane	µg/kg	550[g]	Dyer et al. (2000)
Total chlordane[h]	µg/kg	550	Dyer et al. (2000)
4,4′-DDD	µg/kg	54	estimated from AWQC and BCF
4,4′-DDE	µg/kg	1,000	Dyer et al. (2000)

Table 2-2. Fish Tissue SL TRVs Used to Determine Round 2 COPCs

COI	Unit (ww)	SL TRV	Source
4,4′-DDT	µg/kg	470	Dyer et al. (2000)
Total DDTs[e]	µg/kg	290[f]	fifth percentile LOAEL as presented in Appendix B of the Ecological PRE (Windward 2005)
Dieldrin	µg/kg	220	Dyer et al. (2000)
alpha-Endosulfan	µg/kg	73	Dyer et al. (2000)
beta-Endosulfan	µg/kg	73	Dyer et al. (2000)
Total endosulfan[i]	µg/kg	73	Dyer et al. (2000)
Endosulfan sulfate	µg/kg	73	Dyer et al. (2000)
Heptachlor	µg/kg	60	estimated from AWQC and BCF
Heptachlor epoxide	µg/kg	55	estimated from AWQC and BCF
beta-HCH	µg/kg	4.9	estimated from AWQC and BCF
delta-HCH	µg/kg	4.9	estimated from AWQC and BCF
gamma-HCH	µg/kg	23	Dyer et al. (2000)
Methoxychlor	µg/kg	200	Dyer et al. (2000)
cis-Nonachlor	µg/kg	550[g]	Dyer et al. (2000)
trans-Nonachlor	µg/kg	550[g]	Dyer et al. (2000)
Oxychlordane	µg/kg	550[g]	Dyer et al. (2000)

[a] The SL TRV was based on the fifth percentile LOAEL data reviewed by LWG; and per EPA's comments (EPA 2006a), this value is "probably adequately conservative for the PRE."

[b] The SL TRV based on chromium as Cr^{3+}.

[c] EPA's comments (EPA 2006a) suggested that additional studies be added to the LWG toxicological dataset for PCBs; however, LWG's datasetincludes all acceptable toxicological studies with endpoints relevant to survival, growth, reproduction, and behaviors directly related to survival, growth, or reproduction. Therefore, no additional studies were added to LWG'sdataset. It should be noted that the fifth percentile LOAEL TRV based on the toxicological data (720 µg/kg) is less than the fifth percentile LOAEL TRV reported by Dyer et al. (2000) (800 µg/kg), so the use of Dyer et al. would result in a higher fish tissue SL TRV.

[d] EPA did not propose a change to the selected SL TRV for 2,3,7,8-TCDD (EPA 2006a).

[e] Total DDT is a sum that includes the following COIs: 2,4′-DDD, 2,4′-DDE, 2,4′-DDT, 4,4′-DDD, 4,4′-DDE, and 4,4′-DDT.

[f] The SL TRV was based on the fifth percentile LOAEL data reviewed by LWG; and per EPA's comments (EPA 2006a), this value is "perhaps adequately conservative."

[g] The SL TRV for total chlordane was used as a surrogate.

[h] Total chlordane is a sum that includes the following COIs: cis-chlordane, trans-chlordane, oxychlordane, cis-nonachlor, and trans-nonachlor.

[i] Total endosulfan is a sum that includes the following COIs: alpha-endosulfan, gamma-endosulfan, and endosulfan sulfate.

AWQC – ambient water quality criteria
BCF – bioconcentration factor
BEHP – bis(2-ethylhexyl) phthalate
COI – chemical of interest
COPC – chemical of potential concern
HCH – hexachlorocyclohexane
LC50 – concentration that is lethal to 50% of an exposed population
LOAEL – lowest-observed-adverse-effect level
LWG – Lower Willamette Group
PCB – polychlorinated biphenyl
SL – screening level
SVOC – semivolatile organic compound
TCDD – tetrachlorodibenzo-*p*-dioxin
TRV – toxicity reference value
ww – wet weight

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Table 2-3. Results of Round 2 COPC Screen for Largescale Sucker Tissue

COI	Unit	Maximum Tissue Concentration	SL TRV	Round 2 COPC?[a]
Metals				
Antimony	mg/kg	0.003 J	0.03	no
Arsenic	mg/kg	0.27	1.7	no
Cadmium	mg/kg	0.0325 JT	0.09	no
Chromium	mg/kg	2.77 T	2.7	**yes**
Copper	mg/kg	1.1 T	3.1	no
Lead	mg/kg	0.191 T	2.2	no
Mercury	mg/kg	0.085	0.46	no
Nickel	mg/kg	0.81	18.4	no
Selenium	mg/kg	0.3 U	1.1	no
Silver	mg/kg	0.0074 UJ	0.27	no
Thallium	mg/kg	0.0038 J	4.6	no
Zinc	mg/kg	19.7 T	27	no
Butyltins				
Tributyltin	µg/kg	NA	49.9	NC
Phthalates				
BEHP	µg/kg	3,000 JT	390	**yes**
Butylbenzyl phthalate	µg/kg	330 U	1,200	no
Dibutyl phthalate	µg/kg	520 U	270	no[a]
Di-n-octyl phthalate	µg/kg	1,100 U	41,000	no
SVOCs				
Dibenzofuran	µg/kg	39 UT	1000	no
Hexachlorobenzene	µg/kg	32 UT	490	no
Hexachlorobutadiene	µg/kg	4.6 UT	26	no
Hexachloroethane	µg/kg	33 UT	47,000	no
Phenols				
4-Methylphenol	µg/kg	33 UT	76,500	no
Pentachlorophenol	µg/kg	170 UT	3,100	no
Phenol	µg/kg	330 U	6,200	no
PCBs				
Total PCBs	µg/kg	2,060 J	720	**yes**
Dioxins and Furans				
2,3,7,8-TCDD	pg/g	NA	90	NC
Pesticides				
4,4′-DDD	µg/kg	150 T	54	**yes**
4,4′-DDE	µg/kg	185 T	1,000	no
4,4′-DDT	µg/kg	245 T	470	no
Total DDTs	µg/kg	673	290	**yes**
Aldrin	µg/kg	13 UT	810	no
cis-Chlordane	µg/kg	20 U	550	no

Table 2-3. Results of Round 2 COPC Screen for Largescale Sucker Tissue

COI	Unit	Maximum Tissue Concentration	SL TRV	Round 2 COPC?[a]
trans-Chlordane	µg/kg	8.4 UT	550	no
Total chlordane	µg/kg	24.1 NJ	550	no
Dieldrin	µg/kg	14 UT	220	no
alpha-Endosulfan	µg/kg	20 U	73	no
beta-Endosulfan	µg/kg	15 UT	73	no
Total endosulfan	µg/kg	20 U	73	no
Endosulfan sulfate	µg/kg	12 UT	73	no
Heptachlor	µg/kg	13 UT	60	no
Heptachlor epoxide	µg/kg	8 UT	55	no
beta-HCH	µg/kg	8.5 UT	4.9	no[b]
delta-HCH	µg/kg	7.3 UT	4.9	no[b]
gamma-HCH	µg/kg	9.6 UT	23	no
cis-Nonachlor	µg/kg	17 U	550	no
trans-Nonachlor	µg/kg	11 UT	550	no
Methoxychlor	µg/kg	4.8 UT	200	no
Oxychlordane	µg/kg	20 UT	550	no

[a] Round 2 COPCs were identified where maximum detected concentration in tissue was greater than SL TRV.

[b] Maximum value is a non-detected concentration; COI was not detected in tissue and therefore COI was not retained as a COI.

BEHP – bis(2-ethylhexyl) phthalate
COI – chemical of interest
COPC – chemical of potential concern
HCH – hexachlorocyclohexane
J – estimated concentration
N – presumptive evidence of a compound
NA – not available
NC – not calculated
PCB – polychlorinated biphenyl
SL – screening level
SVOC – semivolatile organic compound
T – value calculated or selected from multiple results
TRV – toxicity reference value
TCDD – tetrachlorodibenzo-*p*-dioxin
U – not detected at given concentration**Bold** identifies maximum detected concentration in tissue was greater than SL TRV.

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Table 2-4. Results of Round 2 COPC Screen for Carp Tissue

COI	Unit	Maximum Tissue Concentration	SL TRV	Round 2 COPC?[a]
2,3,7,8-TCDD	pg/g	1.07	90	no

[a] Round 2 COPCs were identified where maximum detected concentration in tissue was greater than SL TRV.

COI – chemical of interest
COPC – chemical of potential concern
SL – screening level
TCDD – tetrachlorodibenzo-*p*-dioxin
TRV – toxicity reference value

Table 2-5. Results of Round 2 COPC Screen for Sculpin Tissue

COI	Unit	Maximum Tissue Concentration	SL TRV	Round 2 COPC?[a]
Metals				
Antimony	mg/kg	0.009 U	0.03	no
Arsenic	mg/kg	0.3	1.7	no
Cadmium	mg/kg	0.022	0.09	no
Chromium	mg/kg	0.35	2.7	no
Copper	mg/kg	1.68	3.1	no
Lead	mg/kg	0.96 J	2.2	no
Mercury	mg/kg	0.086	0.46	no
Nickel	mg/kg	0.398 J	18.4	no
Selenium	mg/kg	0.3	1.1	no
Silver	mg/kg	0.0067	0.27	no
Thallium	mg/kg	0.012	4.6	no
Zinc	mg/kg	18	27	no
Butyltins				
Tributyltin	µg/kg	NA	49.9	NC
Phthalates				
Bis(2-ethylhexyl) phthalate	µg/kg	28,000 JT	390	**yes**
Butylbenzyl phthalate	µg/kg	330 U	1,200	no
Dibutyl phthalate	µg/kg	330 U	270	no[b]
Di-n-octyl phthalate	µg/kg	600 J	41,000	no
SVOCs				
Dibenzofuran	µg/kg	33 T	1,000	no
Hexachlorobenzene	µg/kg	33 UT	490	no
Hexachlorobutadiene	µg/kg	33 UT	26	no[c]
Hexachloroethane	µg/kg	33 UT	47,000	no
Phenols				
4-Methylphenol	µg/kg	62 T	76,500	no
Pentachlorophenol	µg/kg	170 UT	3,100	no
Phenol	µg/kg	330 U	6,200	no
PCBs				
Total PCBs	µg/kg	3,400	720	**yes**
Dioxins and Furans				
2,3,7,8-TCDD	pg/g	0.446 T	90	no
Pesticides				
4,4′-DDD	µg/kg	305 T	54	**yes**
4,4′-DDE	µg/kg	630 T	1,000	no
4,4′-DDT	µg/kg	1,700 T	470	**yes**
Total DDTs	µg/kg	3,060	290	**yes**
Aldrin	µg/kg	13 UT	810	no

Table 2-5. Results of Round 2 COPC Screen for Sculpin Tissue

COI	Unit	Maximum Tissue Concentration	SL TRV	Round 2 COPC?[a]
cis-Chlordane	µg/kg	20 U	550	no
trans-Chlordane	µg/kg	9.6 U	550	no
Total chlordane	µg/kg	23.2 NJ	550	no
Dieldrin	µg/kg	24 JT	220	no
alpha-Endosulfan	µg/kg	20 U	73	no
beta-Endosulfan	µg/kg	15 UT	73	no
Total Endosulfan	µg/kg	20 U	73	no
Endosulfan sulfate	µg/kg	12 UT	73	no
Heptachlor	µg/kg	13 UT	60	no
Heptachlor epoxide	µg/kg	9.6 U	55	no
beta-HCH	µg/kg	9.6 U	4.9	no[d]
delta-HCH	µg/kg	9.6 U	4.9	no[d]
gamma-HCH	µg/kg	9.6 U	23	no
Methoxychlor	µg/kg	9.6 U	200	no
cis-Nonachlor	µg/kg	11 U	550	no
trans-Nonachlor	µg/kg	11 UT	550	no
Oxychlordane	µg/kg	20 UT	550	no

[a] Round 2 COPCs were identified where maximum detected concentration in tissue was greater than SL TRV.

[b] Maximum value is a non-detected concentration; COI was not detected in tissue and therefore, COI was not retained as a COI.

[c] Maximum value is a non-detected concentration and maximum detected concentration exceeds the SL TRV.

[d] Maximum value is a non-detected concentration; however, maximum detected concentrations did not exceed the SL TRV and therefore, COI was not retained as a COI.

COI – chemical of interest
COPC – chemical of potential concern
HCH – hexachlorocyclohexane
J – estimated concentration
N – presumptive evidence of a compound
NA – not available
NC – not calculated
PCB – polychlorinated biphenyl
SL – screening level
SVOC – semivolatile organic compound
T – value calculated or selected from multiple results
TCDD – tetrachlorodibenzo-*p*-dioxin
TRV – toxicity reference value
U – not detected at given concentration
Bold identifies where maximum detected concentration in tissue was greater than SL TRV.

Table 2-6. Results of Round 2 COPC Screen for Peamouth Tissue

COI	Unit	Maximum Tissue Concentration	SL TRV	Round 2 COPC?[a]
Metals				
Antimony	mg/kg	0.005 U	0.03	no
Arsenic	mg/kg	0.48	1.7	no
Cadmium	mg/kg	0.053	0.09	no
Chromium	mg/kg	0.49	2.7	no
Copper	mg/kg	1.61	3.1	no
Lead	mg/kg	10.6	2.2	**yes**
Mercury	mg/kg	0.054	0.46	no
Nickel	mg/kg	0.482 J	18.4	no
Selenium	mg/kg	0.4	1.1	no
Silver	mg/kg	0.0043 U	0.27	no
Thallium	mg/kg	0.0093	4.6	no
Zinc	mg/kg	25.2	27	no
Butyltins				
Tributyltin	µg/kg	NA	49.9	NC
Phthalates				
Bis(2-ethylhexyl) phthalate	µg/kg	NA	390	NC
Butylbenzyl phthalate	µg/kg	NA	1,200	NC
Dibutyl phthalate	µg/kg	NA	270	NC
Di-n-octyl phthalate	µg/kg	NA	41,000	NC
SVOCs				
Dibenzofuran	µg/kg	NA	1,000	NC
Hexachlorobenzene	µg/kg	7.3 NT	490	no
Hexachlorobutadiene	µg/kg	1 UJT	26	no
Hexachloroethane	µg/kg	3.7 NJT	47,000	no
Phenols				
4-Methylphenol	µg/kg	NA	76,500	NC
Pentachlorophenol	µg/kg	NA	3,100	NC
Phenol	µg/kg	NA	6,200	NC
PCBs				
Total PCBs	µg/kg	300	720	no
Dioxins and Furans				
2,3,7,8-TCDD	pg/g	NA	90	NC
Pesticides				
4,4′-DDD	µg/kg	29.5 T	54	no
4,4′-DDE	µg/kg	185 T	1,000	no
4,4′-DDT	µg/kg	7 T	470	no
Total DDTs	µg/kg	225	290	no
Aldrin	µg/kg	1.7 UT	810	no

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Table 2-6. Results of Round 2 COPC Screen for Peamouth Tissue

COI	Unit	Maximum Tissue Concentration	SL TRV	Round 2 COPC?[a]
cis-Chlordane	µg/kg	3.4 N	550	no
trans-Chlordane	µg/kg	5.6 UT	550	no
Total Chlordane	µg/kg	15.5 N	550	no
Dieldrin	µg/kg	3.3 UT	220	no
alpha-Endosulfan	µg/kg	1 U	73	no
beta-Endosulfan	µg/kg	1.2 UT	73	no
Total Endosulfan	µg/kg	1.2 U	73	no
Endosulfan sulfate	µg/kg	1 UT	73	no
Heptachlor	µg/kg	2.3 UT	60	no
Heptachlor epoxide	µg/kg	1 UT	55	no
beta-HCH	µg/kg	4.1 UT	4.9	no
delta-HCH	µg/kg	2.2 UT	4.9	no
gamma-HCH	µg/kg	3.3 UT	23	no
Methoxychlor	µg/kg	4.8 UT	200	no
cis-Nonachlor	µg/kg	7.1 UT	550	no
trans-Nonachlor	µg/kg	11 UT	550	no
Oxychlordane	µg/kg	1 UT	550	no

[a] Round 2 COPCs were identified where maximum detected concentration in tissue was greater than SL TRV.

COI – chemical of interest
COPC – chemical of potential concern
HCH – hexachlorocyclohexane
J – estimated concentration
N – presumptive evidence of a compound
NA – no tissue chemistry data were available
NC – not calculated; no tissue chemistry data were available
PCB – ploychorinated biphenyl
SL – screening level
SVOC – semivilitile organic compound
T – value calculated or selected from multiple results
TCDD – tetrachlorodibenzo-*p*-dioxin
TRV – toxicity reference value
U – not detected at given concentration

Table 2-7. Results of Round 2 COPC Screen for Juvenile Chinook Salmon Tissue

COI	Unit	Maximum Tissue Concentration	SL TRV	Round 2 COPC?[a]
Metals				
Antimony	mg/kg	0.00429 U	0.03	no
Arsenic	mg/kg	0.25	1.7	no
Cadmium	mg/kg	0.027	0.09	no
Chromium	mg/kg	0.19	2.7	no
Copper	mg/kg	2.15	3.1	no
Lead	mg/kg	0.0741 U	2.2	no
Mercury	mg/kg	0.02	0.46	no
Nickel	mg/kg	0.492 J	18.4	no
Selenium	mg/kg	0.4 U	1.1	no
Silver	mg/kg	0.002992 U	0.27	no
Thallium	mg/kg	0.0105	4.6	no
Zinc	mg/kg	33.3	27	**yes**
Butyltins				
Tributyltin ion	µg/kg	4.1	49.9	no
Phthalates				
Bis(2-ethylhexyl) phthalate	µg/kg	860 UT	390	no[b]
Butylbenzyl phthalate	µg/kg	3700 U	1,200	no[c]
Dibutyl phthalate	µg/kg	520 U	270	no[c]
Di-n-octyl phthalate	µg/kg	560 U	41,000	no
SVOCs				
Dibenzofuran	µg/kg	33 UT	1,000	no
Hexachlorobenzene	µg/kg	33 UT	490	no
Hexachlorobutadiene	µg/kg	1.1 U	26	no
Hexachloroethane	µg/kg	120 U	47,000	no
Phenols				
4-Methylphenol	µg/kg	210 U	76,500	no
Pentachlorophenol	µg/kg	420 U	3,100	no
Phenol	µg/kg	330 U	6,200	no
PCBs				
Total PCBs	µg/kg	277 J	720	no
Dioxins and Furans				
2,3,7,8-TCDD	pg/g	0.876	90	no
Pesticides				
4,4′-DDD	µg/kg	130	54	**yes**
4,4′-DDE	µg/kg	93	1,000	no
4,4′-DDT	µg/kg	38	470	no
Total DDTs	µg/kg	285	290	no

Table 2-7. Results of Round 2 COPC Screen for Juvenile Chinook Salmon Tissue

COI	Unit	Maximum Tissue Concentration	SL TRV	Round 2 COPC?[a]
Aldrin	µg/kg	1.2 U	810	no
cis-Chlordane	µg/kg	1.6 N	550	no
trans-Chlordane	µg/kg	4	550	no
Total chlordane	µg/kg	19 U	550	no
Dieldrin	µg/kg	2.7 U	220	no
alpha-Endosulfan	µg/kg	1.1 U	73	no
beta-Endosulfan	µg/kg	19 UT	73	no
Total Endosulfan	µg/kg	19 U	73	no
Endosulfan sulfate	µg/kg	1 J	73	no
Heptachlor	µg/kg	1.2	60	no
Heptachlor epoxide	µg/kg	2.1 NJ	55	no
beta-HCH	µg/kg	1.2 UT	4.9	no
delta-HCH	µg/kg	1.1 U	4.9	no
gamma-HCH	µg/kg	1.1 U	23	no
Methoxychlor	µg/kg	19 U	200	no
cis-Nonachlor	µg/kg	17 U	550	no
trans-Nonachlor	µg/kg	19 UT	550	no
Oxychlordane	µg/kg	1 U	550	no

[a] Round 2 COPCs were identified where maximum detected concentration in tissue was greater than SL TRV.

[b] Maximum value is a non-detected concentration; COI was not detected in tissue and therefore, COI was not retained as a COI.

[c] Maximum value is a non-detected concentration; however, maximum detected concentrations did not exceed the SL TRV and therefore, COI was not retained as a COI.

COI – chemical of interest

COPC – chemical of potential concern
HCH – hexachlorocyclohexane
J – estimated concentration
N – presumptive evidence of a compound
PCB – polychlorinated biphenyl
SL – screening level
SVOC – semivolitile organic compound
T – value calculated or selected from multiple results
TCDD – tetrachlorodibenzo-*p*-dioxin
TRV – toxicity reference value
U – not detected at given concentration
Bold identifies where maximum detected concentration in tissue was greater than SL TRV.

Table 2-8. Results of Round 2 COPC Screen for Smallmouth Bass Tissue

COI	Unit	Maximum TissueConcentrationi	SL TRV	Round 2 COPC?[a]
Metals				
Antimony	mg/kg	0.004 U	0.03	no
Arsenic	mg/kg	0.39	1.7	no
Cadmium	mg/kg	0.024 J	0.09	no
Chromium	mg/kg	1.14	2.7	no
Copper	mg/kg	1.29	3.1	no
Lead	mg/kg	0.303 J	2.2	no
Mercury	mg/kg	0.114	0.46	no
Nickel	mg/kg	0.2	18.4	no
Selenium	mg/kg	0.4	1.1	no
Silver	mg/kg	0.007 U	0.27	no
Thallium	mg/kg	0.0085	4.6	no
Zinc	mg/kg	16.3	27	no
Butyltins				
Tributyltin	µg/kg	NA	49.9	NC
Phthalates				
Bis(2-ethylhexyl) phthalate	µg/kg	87,000 JT	390	**yes**
Butylbenzyl phthalate	µg/kg	330 U	1,200	no
Dibutyl phthalate	µg/kg	330 U	270	no[b]
Di-n-octyl phthalate	µg/kg	2,100 T	41,000	no
SVOCs				
Dibenzofuran	µg/kg	52 T	1,000	no
Hexachlorobenzene	µg/kg	6.4 UT	490	no
Hexachlorobutadiene	µg/kg	4.6 UT	26	no
Hexachloroethane	µg/kg	13 UT	47,000	no
Phenols				
4-Methylphenol	µg/kg	33 UT	76,500	no
Pentachlorophenol	µg/kg	160 UT	3,100	no
Phenol	µg/kg	330 U	6,200	no
PCBs				
Total PCBs	µg/kg	4,950 J	720	**yes**
Dioxins and Furans				
2,3,7,8-TCDD	pg/g	1.49	90	no
Pesticides				
4,4′-DDD	µg/kg	110 T	54	**yes**
4,4′-DDE	µg/kg	220 T	1,000	no
4,4′-DDT	µg/kg	130 T	470	no
Total DDTs	µg/kg	416	290	**yes**
Aldrin	µg/kg	13 UT	810	no

Table 2-8. Results of Round 2 COPC Screen for Smallmouth Bass Tissue

COI	Unit	Maximum TissueConcentrationi	SL TRV	Round 2 COPC?[a]
cis-Chlordane	µg/kg	20 U	550	no
trans-Chlordane	µg/kg	8.4 UT	550	no
Total chlordane	µg/kg	32 U	550	no
Dieldrin	µg/kg	20 UT	220	no
alpha-Endosulfan	µg/kg	20 U	73	no
beta-Endosulfan	µg/kg	20 UT	73	no
Total Endosulfan	µg/kg	28.4 J	73	no
Endosulfan sulfate	µg/kg	20 UT	73	no
Heptachlor	µg/kg	13 UT	60	no
Heptachlor epoxide	µg/kg	8 UT	55	no
beta-HCH	µg/kg	8.5 UT	4.9	no[b]
delta-HCH	µg/kg	7.3 UT	4.9	no[b]
gamma-HCH	µg/kg	9.6 UT	23	no
Methoxychlor	µg/kg	20 UT	200	no
cis-Nonachlor	µg/kg	7.1 UT	550	no
trans-Nonachlor	µg/kg	11 UT	550	no
Oxychlordane	µg/kg	32 UT	550	no

[a] Round 2 COPCs were identified where maximum detected concentration in tissue was greater than SL TRV.

[b] Maximum value is a non-detected concentration; COI was not detected in tissue and therefore, COI was not retained as a COI.

COI – chemical of interest
COPC – chemical of potential concern
HCH – hexachlorocyclohexane
J – estimated concentration
NA – not available
NC – not calculated
PCB – polychlorinated biphenyl
SL – screening level
SVOC – semivolitile organic compound
T – value calculated or selected from multiple results
TCDD – tetrachlorodibenzo-*p*-dioxin
TRV – toxicity reference value
U – not detected at given concentration
Bold identifies where maximum detected concentration in tissue was greater than SL TRV.

Table 2-9. Results of Round 2 COPC Screen for Northern Pikeminnow Tissue

COI	Unit	Maximum Tissue Concentration	SL TRV	Round 2 COPC?[a]
Metals				
Antimony	mg/kg	0.002 U	0.03	no
Arsenic	mg/kg	0.36	1.7	no
Cadmium	mg/kg	0.012 J	0.09	no
Chromium	mg/kg	0.67	2.7	no
Copper	mg/kg	0.89	3.1	no
Lead	mg/kg	0.016	2.2	no
Mercury	mg/kg	0.494	0.46	**yes**
Nickel	mg/kg	0.461 J	18.4	no
Selenium	mg/kg	0.4	1.1	no
Silver	mg/kg	0.0052 U	0.27	no
Thallium	mg/kg	0.004 J	4.6	no
Zinc	mg/kg	20	27	no
Butyltins				
Tributyltin	µg/kg	NA	49.9	NC
Phthalates				
Bis(2-ethylhexyl) phthalate	µg/kg	NA	390	NC
Butylbenzyl phthalate	µg/kg	NA	1,200	NC
Dibutyl phthalate	µg/kg	NA	270	NC
Di-n-octyl phthalate	µg/kg	NA	41,000	NC
SVOCs				
Dibenzofuran	µg/kg	NA	1,000	NC
Hexachlorobenzene	µg/kg	9.8 U	490	no
Hexachlorobutadiene	µg/kg	9.8 UJ	26	no
Hexachloroethane	µg/kg	13 UT	47,000	no
Phenols				
4-Methylphenol	µg/kg	NA	76,500	NC
Pentachlorophenol	µg/kg	NA	3,100	NC
Phenol	µg/kg	NA	6,200	NC
PCBs				
Total PCBs	µg/kg	1,930	720	**yes**
Dioxins and Furans				
2,3,7,8-TCDD	pg/g	NA	90	NC
Pesticides				
4,4′-DDD	µg/kg	48 T	54	no
4,4′-DDE	µg/kg	545 T	1,000	no
4,4′-DDT	µg/kg	53 J	470	no
Total DDTs	µg/kg	764	290	**yes**
Aldrin	µg/kg	13 UT	810	no

Table 2-9. Results of Round 2 COPC Screen for Northern Pikeminnow Tissue

COI	Unit	Maximum Tissue Concentration	SL TRV	Round 2 COPC?[a]
cis-Chlordane	µg/kg	19 U	550	no
trans-Chlordane	µg/kg	9.8 U	550	no
Total chlordane	µg/kg	19 U	550	no
Dieldrin	µg/kg	14 UT	220	no
alpha-Endosulfan	µg/kg	19 U	73	no
beta-Endosulfan	µg/kg	15 UT	73	no
Total Endosulfan	µg/kg	19 U	73	no
Endosulfan sulfate	µg/kg	12 UT	73	no
Heptachlor	µg/kg	13 UT	60	no
Heptachlor epoxide	µg/kg	9.8 U	55	no
beta-HCH	µg/kg	9.8 U	4.9	no[b]
delta-HCH	µg/kg	9.8 U	4.9	no[b]
gamma-HCH	µg/kg	9.8 U	23	no
Methoxychlor	µg/kg	17 JT	200	no
cis-Nonachlor	µg/kg	9.8 U	550	no
trans-Nonachlor	µg/kg	14 U	550	no
Oxychlordane	µg/kg	19 UT	550	no

[a] Round 2 COPCs were identified where maximum detected concentration in tissue was greater than SL TRV.

[a] Maximum value is a non-detected concentration; COI was not detected in tissue and therefore, COI was not retained as a COI.

COI – chemical of interest
COPC – chemical of potential concern
HCH – hexachlorocyclohexane
J – estimated concentration
PCB – polychlorinated biphenyl
NA – not available
NC – not calculated
SL – screening level
SVOC – semivolatile organic compound
TRV – toxicity reference value
T – value calculated or selected from multiple results
TCDD – tetrachlorodibenzo-*p*-dioxin
U – not detected at given concentration
Bold identifies where maximum detected concentration in tissue was greater than SL TRV.

Table 2-10. Summary of Tissue Round 2 COPC Screen for Fish

COPC	Largescale Sucker	Sculpin	Peamouth	Juvenile Chinook Salmon	Smallmouth Bass	Northern Pikeminnow
Chromium	X					
Lead			X			
Mercury						X
Zinc				X		
BEHP	X	X			X	
Total PCBs	X	X			X	X
4,4′-DDD	X	X		X	X	
4,4′-DDT		X				
beta-HCH		X				
Total DDTs	X	X			X	X

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
HCH – hexachlorocyclohexane
PCB – polychlorinated biphenyl

Table 3-1. Results of Round 2 COPC Predicted Tissue Screen for Sculpin

COI	Unit (ww)	Sediment-Tissue Relationship?	Maximum Predicted Tissue Concentration[a]	SL TRV	Round 2 COPC?[b]
Metals					
Antimony	mg/kg	NA	NA	0.03	NA
Arsenic	mg/kg	no	NA	1.7	NA
Cadmium	mg/kg	yes	0.0124	0.09	no
Chromium	mg/kg	no	NA	2.7	NA
Copper	mg/kg	no	NA	3.1	NA
Lead	mg/kg	yes	0.177	2.2	no
Mercury	mg/kg	yes	0.044	0.46	no
Nickel	mg/kg	no	NA	18.4	NA
Selenium	mg/kg	yes	9.75	1.10	**yes**
Silver	mg/kg	yes	0.0035	0.27	no
Zinc	mg/kg	yes	16.4	27.0	no
Phthalates					
BEHP	mg/kg lipid[c]	yes	1,090	9.35	**yes**
Butylbenzyl phthalate	mg/kg lipid[c]	NA	NA	28.8	NA
Dibutyl phthalate	mg/kg lipid[c]	NA	NA	6.47	NA
Di-n-octyl phthalate	mg/kg lipid[c]	yes	0.931	982	no
SVOCs					
Benzyl alcohol	mg/kg lipid[c]	NA	NA		NA
Dibenzofuran	mg/kg lipid[c]	yes	2.85	24.0	no
Hexachlorobenzene	mg/kg lipid[c]	NA	NA	11.7	NA
Hexachlorobutadiene	mg/kg lipid[c]	yes	0.00675	0.62	no
Hexachloroethane	mg/kg lipid[c]	NA	NA	1126	NA
Phenols					
4-Methylphenol	mg/kg lipid[c]	yes	9.91	1833	no
Pentachlorophenol	mg/kg lipid[c]	NA	NA	74.3	NA
Phenol	mg/kg lipid[c]	NA	NA	149	NA
Pesticides					
Aldrin	mg/kg lipid[c]	NA	NA	19.41	NA
alpha-Endosulfan	mg/kg lipid[c]	NA	NA	1.75	NA
beta-Endosulfan	mg/kg lipid[c]	NA	NA	1.75	NA
beta-HCH	mg/kg lipid[c]	NA	NA	0.12	NA
cis-Chlordane	mg/kg lipid[c]	NA	NA	13.18	NA
cis-Nonachlor	mg/kg lipid[c]	NA	NA	13.18	NA
delta-HCH	mg/kg lipid[c]	NA	NA	0.12	NA
Dieldrin	mg/kg lipid[c]	NA	NA	5.27	NA
Endosulfan sulfate	mg/kg lipid[c]	NA	NA	1.75	NA
gamma-HCH	mg/kg lipid[c]	yes	0.00117	0.55	no
Heptachlor	mg/kg lipid[c]	NA	NA	1.44	NA

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COI	Unit (ww)	Sediment-Tissue Relationship?	Maximum Predicted Tissue Concentration[a]	SL TRV	Round 2 COPC?[b]
Heptachlor epoxide	mg/kg lipid[c]	NA	NA	1.32	NA
Methoxychlor	mg/kg lipid[c]	NA	NA	4.79	NA
Oxychlordane	mg/kg lipid[c]	NA	NA	13.18	NA
Total Chlordane	mg/kg lipid[c]	NA	NA	13.18	NA
Total Endosulfan	mg/kg lipid[c]	yes	0.0501	1.75	no
trans-Chlordane	mg/kg lipid[c]	NA	NA	13.18	NA
trans-Nonachlor	mg/kg lipid[c]	NA	NA	13.18	NA

[a] Predicted tissue concentration is based on the site-specific BSAF and the 95th percentile site-wide sediment concentration.

[b] Round 2 COPCs were identified when the predicted tissue concentration was greater than the SL TRV.

[c] The average sculpin lipid value, 4.17%, was used to estimate lipid-normalized tissue concentrations.

COI – chemicals of interest

COPC –chemical of potential concern

HCH –hexachlorocyclohexane

Bold identifies where maximum predicted concentration in tissue was greater than SL TRV.

Table 4-1. Results of TRV Search for Fish Dietary COIs

Dietary COIs with TRVs	
Arsenic	Silver
Cadmium	Zinc
Copper	Benzo(a)pyrene
Lead	Total PAHs
Mercury	Total DDTs
Selenium	Total PCBs
Dietary COIs with no TRVs	
Antimony	Nickel
Chromium	Thallium

COI – chemical of interest
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TRV – toxicity reference value

Table 4-2. Uncertainty Factors Used to Derive Chronic NOAEL TRVs for Fish

Source	**cLOAEL: cNOAEL**	**sLOAEL: cNOAEL**	**aLOAEL (non-lethal): cNOAEL**	**aLOAEL (lethal; LC50): cNOAEL**
EPA Region 10 Guidance (EPA 1997b)	5	10[a]	10[a]	50

[a] For acute or subchronic LOAEL to chronic NOAEL.
aLOAEL – acute LOAEL
cLOAEL – chronic LOAEL
cNOAEL – chronic NOAEL
sLOAEL – subchronic LOAEL
LC50 – concentration that is lethal to 50% of an exposed population

Table 4-3. Fish Dietary NOAEL TRVs Used to Identify Screening Dietary Dose Round 2 COPCs

COI	Test Species	Unit (bw/day)	NOAEL	Endpoint	Source
Metals					
Arsenic	rainbow trout	mg/kg	0.40	growth	Oladimeji et al. (1984)
Cadmium	rockfish	mg/kg	0.0020[a]	growth	Kang et al. (2005); Kim et al. (2004)
Copper	channel catfish	mg/kg	0.24	growth	Murai et al. (1981)
Lead	rainbow trout	mg/kg	134	growth	Goettl et al. (1976)
Mercury	mummichog	mg/kg	0.013	reduced survival	Matta et al. (2001)
Selenium	bluegill	mg/kg	0.10	reduced survival	Cleveland et al. (1993)
Silver	rainbow trout	mg/kg	70	growth	Galvez and Wood (1999)
Zinc	rainbow trout	mg/kg	19	growth	Takeda and Shimma (1977)
Butyltins					
TBT	Japanese flounder	µg/kg	2.1	growth, sex reversal	Shimasaki et al. (2003)
PAHs					
Benzo(a)pyrene	English sole	µg/kg	660	growth	Hart and Heddle (1991)
Total PAHs	chinook salmon	µg/kg	6,100	growth, disease resistance	Meador et al. (2006)
PCBs					
Total PCBs	barbel	µg/kg	10[a]	reproduction	Hugla and Thome (1999)
Pesticides					
Total DDTs	brook trout	µg/kg	28[a]	reproduction	Macek (1968)

[a] NOAEL was extrapolated from LOAEL using a UF of 5.
bw – body weight
COI – chemical of interest
COPC – chemical of potential concern
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TBT – tributyltin
TRV – toxicity reference value

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Table 4-4. Prey Species Identified for Each Fish Receptor for Screening COIs

Receptor Species	Prey Item
Largescale sucker	worm, clam[a]
White sturgeon (pre-breeding)	worm, clam[a]
Juvenile chinook salmon	multiplate invertebrate tissue,[b] and stomach contents (Round 2)
Sculpin	worm, clam,[a] sculpin[c]
Peamouth	multiplate invertebrate tissue,[b,c] worm, clam,[a] sculpin[c]
Smallmouth bass	sculpin,[c] crayfish,[c] worm
Northern pikeminnow	crayfish,[c] sculpin,[c] worm, largescale sucker,[c] carp,[c] peamouth,[c] northern pikeminnow[c]

[a] Field-collected clam tissue concentrations were used to represent clam prey tissue.
[b] Laboratory-exposed worm concentrations of PAHs, mercury, and TBT were used as a surrogate species for multiplates invertebrate tissue.
[c] Because TBT was only analyzed for in clams and laboratory worms, laboratory worm data were conservatively used as a surrogate for all other prey because TBT concentrations were higher in worm tissue.
COI – chemical of interest

Table 4-5. Exposure Parameters Used for Screening Fish Dietary COIs

Receptor	BW (kg)	FIR (kg ww/ day)	Moisture in Diet (%)	SI (%)[a]	SIR (kg dw/day)[a]
Largescale sucker	0.794	0.0332	85	8	0.00398
White sturgeon (pre-breeding)	15.9[b]	0.663[b]	85	56	0.0557
Juvenile chinook salmon	0.012	0.000501	85	1	0.00000105
Sculpin	0.019	0.000794	79	5	0.00000834
Peamouth	0.104	0.00435	79	5	0.0000456
Smallmouth bass	0.372[c]	0.0155[c]	74	1	0.0000404
Northern pikeminnow	0.558	0.0233	74	1	0.0000606

[a] Sediment ingestion and sediment ingestion rates for fish receptors have been updated from the values presented in the Ecological PRE (Windward 2005) to reflect more realistic portions of incidental sediment ingestion based on discussions with fisheries experts.
[b] Dietary dose assessment for pre-breeding sturgeon has been added since the submittal of the Ecological PRE (Windward 2005). Body weight was estimated using an allometric model, incorporating the range of sturgeon body lengths targeted for Round 3 tissue collection.
[c] Body weight and food ingestion rate have been updated from the Ecological PRE to reflect smallmouth bass collected only from within the Study Area (RM 2 to RM 11).
COI – chemical of interest
BW – body weight
dw – dry weight
FIR – food ingestion rate
SI – sediment ingestion
SIR – sediment ingestion rate (SIR = FIR * SI)
ww – wet weight

Table 4-6. Results of Round 2 Dietary COPC Screen for Largescale Sucker

COI	Exposure Concentration			Dose Concentration			Round 2 COPC?[a]
	Max C_{prey} (ww)	Max C_{sed} (dw)	Unit	Max IR_{diet}	NOAEL TRV	Unit (bw/day)	
Arsenic	3.04	75.6	mg/kg	0.165	0.4	mg/kg	no
Cadmium	0.254	46.2	mg/kg	0.0338	0.002	mg/kg	**yes**
Copper	20.2	1,080	mg/kg	1.39	0.24	mg/kg	**yes**
Lead	0.847 T	1,950	mg/kg	1.01	134	mg/kg	no
Mercury	0.016 J	4.84	mg/kg	0.00310	0.013	mg/kg	no
Selenium	0.37	20	mg/kg	0.0255	0.10	mg/kg	no
Silver	0.101	14.8	mg/kg	0.0116	70	mg/kg	no
Zinc	54	2,850	mg/kg	3.69	19	mg/kg	no
Tributyltin ion	1,700	47,000	µg/kg	94.6	2.1	µg/kg	**yes**
Benzo(a)pyrene	1,500	340,000	µg/kg	233	660	µg/kg	no
Total PAHs	37,300	7,260,000	µg/kg	5,200	6,100	µg/kg	no
Total PCBs	4,310 J	30,800 J	µg/kg	196	10	µg/kg	**yes**
Total DDTs	1,490	16,200 NJ	µg/kg	70.4	28	µg/kg	**yes**

[a] Round 2 COPCs were identified where the maximum estimated dietary dose was greater than the NOAEL TRV.

bw – body weight
COI – chemical of interest
COPC – chemical of potential concern
dw – dry weight
IR – ingestion rate
J – estimated concentration
N – presumptive evidence of a compound
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
T – value calculated or selected from multiple results
TRV – toxicity reference value
ww – wet weight
Bold identifies where the maximum estimated dietary dose was greater than the NOAEL TRV.

Table 4-7. Results of Round 2 Dietary COPC Screen for Pre-Breeding Sturgeon[a]

COI	Exposure Concentration			Dose Concentration			Round 2 COPC?[a]
	Max C_{prey} (ww)	Max C_{sed} (dw)	Unit	Max IR_{diet}	NOAEL TRV	Unit (bw/day)	
Arsenic	3.04	75.6	mg/kg	0.392	0.40	mg/kg	no
Cadmium	0.254	46.2	mg/kg	0.172	0.002	mg/kg	**yes**
Copper	20.2	1,080	mg/kg	4.63	0.24	mg/kg	**yes**
Lead	0.847 T	1,950	mg/kg	6.87	134	mg/kg	no
Mercury	0.016 J	4.84	mg/kg	0.0176	0.013	mg/kg	**yes**
Selenium	0.37	20	mg/kg	0.0855	0.10	mg/kg	no
Silver	0.101	14.8	mg/kg	0.0561	70	mg/kg	no
Zinc	54	2,850	mg/kg	12.2	19	mg/kg	no
Tributyltin ion	1,700	47,000	µg/kg	236	2.1	µg/kg	**yes**
Benzo(a)pyrene	1,500	340,000	µg/kg	1,250	660	µg/kg	**yes**
Total PAHs	37,300	7,260,000	µg/kg	27,000	6,100	µg/kg	**yes**
Total PCBs	4,310 J	30,800 J	µg/kg	288	10	µg/kg	**yes**
Total DDTs	1,490	16,200 NJ	µg/kg	119	28	µg/kg	**yes**

[a] Dietary dose for pre-breeding sturgeon estimated assuming 56% sediment ingestion.

[b] Round 2 COPCs were identified where the maximum estimated dietary dose was greater than the NOAEL TRV..

bw – body weight
COI – chemical of interest
COPC – chemical of potential concern
dw – dry weight
IR – ingestion rate
J – estimated concentration
N – presumptive evidence of a compound
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
T – value calculated or selected from multiple results
TRV – toxicity reference value
ww – wet weight
Bold identifies where the maximum estimated dietary dose was greater than the NOAEL TRV.

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Table 4-8. Results of Dietary Dose Round 2 COPC Screen for Sculpin

COI	Exposure Concentration			Dose Concentration			Round 2 COPC?[a]
	Max C_{prey} (ww)	Max C_{sed} (dw)	Unit	Max IR_{diet}	NOAEL TRV	Unit (bw/day)	
Arsenic	3.04	75.6	mg/kg	0.160	0.40	mg/kg	no
Cadmium	0.254	46.2	mg/kg	0.0309	0.002	mg/kg	**yes**
Copper	20.2	1,080	mg/kg	1.32	0.24	mg/kg	**yes**
Lead	0.96 J	1,950	mg/kg	0.896	134	mg/kg	no
Mercury	0.086	4.84	mg/kg	0.00572	0.013	mg/kg	no
Selenium	0.37	20	mg/kg	0.0242	0.10	mg/kg	no
Silver	0.101	14.8	mg/kg	0.0107	70	mg/kg	no
Zinc	54	2,850	mg/kg	3.51	19	mg/kg	no
Tributyltin ion	1,700	47,000	µg/kg	91.7	2.1	µg/kg	**yes**
Benzo(a)pyrene	1,500	340,000	µg/kg	212	660	µg/kg	no
Total PAHs	37,300	7,260,000	µg/kg	4746	6,100	µg/kg	no
Total PCBs	4,310 J	30,800 J	µg/kg	194	10	µg/kg	**yes**
Total DDTs	3,060	16,200 NJ	µg/kg	135	28	µg/kg	**yes**

[a] Round 2 COPCs were identified where the maximum estimated dietary dose was greater than the NOAEL TRV.

bw – body weight
COI – chemical of interest
COPC – chemical of potential concern
dw – dry weightIR – ingestion rate
J – estimated concentration
N – presumptive evidence of a compound
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TRV – toxicity reference value
ww – wet weight
Bold identifies where the maximum estimated dietary dose was greater than the NOAEL TRV.

Table 4-9. Results of Dietary Dose Round 2 COPC Screen for Peamouth

COI	Exposure Concentration			Dose Concentration			Round 2 COPC?[a]
	Max C_{prey} (ww)	Max C_{sed} (dw)	Unit	Max IR_{diet}	NOAEL TRV	Unit (bw/day)	
Arsenic	3.04	75.6	mg/kg	0.160	0.40	mg/kg	no
Cadmium	0.254	46.2	mg/kg	0.0309	0.002	mg/kg	**yes**
Copper	20.2	1,080	mg/kg	1.32	0.24	mg/kg	**yes**
Lead	1.06 J	1,950	mg/kg	0.899	134	mg/kg	no
Mercury	0.086	4.84	mg/kg	0.00572	0.013	mg/kg	no
Selenium	0.37	20	mg/kg	0.0242	0.10	mg/kg	no
Silver	0.101	14.8	mg/kg	0.0107	70	mg/kg	no
Zinc	54	2,850	mg/kg	3.51	19	mg/kg	no
Tributyltin ion	1,700	47,000	µg/kg	91.7	2.1	µg/kg	**yes**
Benzo(a)pyrene	1,500	340,000	µg/kg	212	660	µg/kg	no
Total PAHs	37,300	7,260,000	µg/kg	4,743	6,100	µg/kg	no
Total PCBs	4,310 J	30,800 J	µg/kg	194	10	µg/kg	**yes**
Total DDTs	3,060	16,200 NJ	µg/kg	135	28	µg/kg	**yes**

[a] Round 2 COPCs were identified where the maximum estimated dietary dose was greater than the NOAEL TRV.

bw – body weight
COI – chemical of interest
COPC – chemical of potential concern
dw – dry weight
IR – ingestion rate
J – estimated concentration
N – presumptive evidence of a compound
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TRV – toxicity reference value
ww – wet weight
Bold identifies where the maximum estimated dietary dose was greater than the NOAEL TRV.

Table 4-10. Results of Dietary Dose Round 2 COPC Screen for Juvenile Chinook Salmon[a]

COI	Exposure Concentration			Dose Concentration			Round 2 COPC?[a]
	Max C_{prey} (ww)	Max C_{sed} (dw)	Unit	Max IR_{diet}	NOAEL TRV	Unit (bw/day)	
Arsenic	0.45	75.6	mg/kg	0.0254	0.40	mg/kg	no
Cadmium	0.0366	46.2	mg/kg	0.00557	0.002	mg/kg	**yes**
Copper	6.0 J	1,080	mg/kg	0.345	0.24	mg/kg	**yes**
Lead	1.06 J	1,950	mg/kg	0.215	134	mg/kg	no
Mercury	0.0105	4.84	mg/kg	0.000862	0.013	mg/kg	no
Selenium	0.06	20	mg/kg	0.00426	0.10	mg/kg	no
Silver	0.0287 J	14.8	mg/kg	0.00249	70	mg/kg	no
Zinc	24.8 J	2,850	mg/kg	1.28	19	mg/kg	no
Tributyltin ion	1700	47,000	µg/kg	75.1	2.1	µg/kg	**yes**
Benzo(a)pyrene	1,500	340,000	µg/kg	92.4	660	µg/kg	no
Total PAHs	37,300	7,260,000	µg/kg	2193	6,100	µg/kg	no
Total PCBs	498	30,800 J	µg/kg	23.5	10	µg/kg	**yes**
Total DDTs	327 NJ	16,200 NJ	µg/kg	15.1	28	µg/kg	no

[a] Dietary dose for juvenile chinook estimated using modeled prey. Maximum juvenile chinook salmon prey tissue concentrations were equal to or greater than tissue concentrations measured in juvenile chinook salmon stomach contents for those COPCs that were analyzed in stomach contents tissue (i.e., total PAHs, total PCBs, and total DDTs).

[b] Round 2 COPCs were identified where the maximum estimated dietary dose was greater than the NOAEL TRV.

bw – body weight
COI – chemical of interest
COPC – chemical of potential concern
dw – dry weight
IR – ingestion rate
J – estimated concentration
N – presumptive evidence of a compound
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TRV – toxicity reference value
ww – wet weight
Bold identifies where the maximum estimated dietary dose was greater than the NOAEL TRV.

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Table 4-11. Results of Dietary Dose Round 2 COPC Screen for Smallmouth Bass

COI	Exposure Concentration			Dose Concentration			Round 2 COPC?[a]
	Max C_{prey} (ww)	Max C_{sed} (dw)	Unit	Max IR_{diet}	NOAEL TRV	Unit (bw/day)	
Arsenic	3.04	75.6	mg/kg	0.135	0.40	mg/kg	no
Cadmium	0.254	46.2	mg/kg	0.0156	0.002	mg/kg	**yes**
Copper	20.2	1,080	mg/kg	0.959	0.24	mg/kg	**yes**
Lead	1.3 JT	1,950	mg/kg	0.266	134	mg/kg	no
Mercury	0.086	4.84	mg/kg	0.00411	0.013	mg/kg	no
Selenium	0.37	20	mg/kg	0.0176	0.10	mg/kg	no
Silver	0.0472 J	14.8	mg/kg	0.00357	70	mg/kg	no
Zinc	31.5	2,850	mg/kg	1.62	19	mg/kg	no
Tributyltin ion	1,700	47,000	µg/kg	75.9	2.1	µg/kg	**yes**
Benzo(a)pyrene	1,500	340,000	µg/kg	99.4	660	µg/kg	no
Total PAHs	37,300	7,260,000	µg/kg	2,343	6,100	µg/kg	no
Total PCBs	4,310 J	30,800 J	µg/kg	183	10	µg/kg	**yes**
Total DDTs	3,060	16,200 NJ	µg/kg	129	28	µg/kg	**yes**

[a] Round 2 COPCs were identified where the maximum estimated dietary dose was greater than the NOAEL TRV.

bw – body weight
COI – chemical of interest
COPC – chemical of potential concern
dw – dry weight
IR – ingestion rate
J – estimated concentration
N – presumptive evidence of a compound
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
T – value calculated or selected from multiple results
TRV – toxicity reference value
ww – wet weight
Bold identifies where the maximum estimated dietary dose was greater than the NOAEL TRV.

Table 4-12. Results of Dietary Dose Round 2 COPC Screen for Northern Pikeminnow

COI	Exposure Concentration			Dose Concentration			Round 2 COPC?[a]
	Max C_{prey} (ww)	Max C_{sed} (dw)	Unit	Max IR_{diet}	NOAEL TRV	Unit (bw/day)	
Arsenic	3.04	75.6	mg/kg	0.135	0.40	mg/kg	no
Cadmium	0.254	46.2	mg/kg	0.0156	0.002	mg/kg	**yes**
Copper	20.2	1,080	mg/kg	0.961	0.24	mg/kg	**yes**
Lead	10.6	1,950	mg/kg	0.654	134	mg/kg	no
Mercury	0.494	4.84	mg/kg	0.0212	0.013	mg/kg	**yes**
Selenium	0.4	20	mg/kg	0.0189	0.10	mg/kg	no
Silver	0.0472 J	14.8	mg/kg	0.00358	70	mg/kg	no
Zinc	112	2,850	mg/kg	4.99	19	mg/kg	no
Tributyltin ion	1,700	47,000	µg/kg	76.1	2.1	µg/kg	**yes**
Benzo(a)pyrene	1,500	340,000	µg/kg	100	1,900	µg/kg	no
Total PAHs	37,300	7,260,000	µg/kg	2,346	6,100	µg/kg	no
Total PCBs	7,460 J	30,800 J	µg/kg	315	10	µg/kg	**yes**
Total DDTs	3,060	16,200 NJ	µg/kg	130	28	µg/kg	**yes**

[a] Round 2 COPCs were identified where the maximum estimated dietary dose was greater than the NOAEL TRV.

bw – body weight
COI – chemical of interest
COPC – chemical of potential concern
dw – dry weight
IR – ingestion rate
J – estimated concentration
N – presumptive evidence of a compound
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TRV – toxicity reference value
ww – wet weight
Bold identifies where the maximum estimated dietary dose was greater than the NOAEL TRV.

Table 4-13. Summary of Dietary Dose Round 2 COPCs for Fish

Round 2 COPC	Largescale Sucker	Pre-Breeding Sturgeon	Sculpin	Peamouth	Juvenile Chinook Salmon	Smallmouth Bass	Northern Pikeminnow
Cadmium	X	X	X	X	X	X	X
Copper	X	X	X	X	X	X	X
Mercury		X					X
Tributyltin ion	X	X	X	X	X	X	X
Benzo(a) pyrene		X					
Total PAHs		X					
Total PCBs	X	X	X	X	X	X	X
Total DDTs	X	X	X	X		X	X

COPC – chemical of potential concern
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl

Table 5-1. Surface Water COIs for Fish

COIs	
Metals	
Antimony	Nickel
Arsenic	Selenium
Cadmium	Silver
Chromium	Thallium
Copper	Zinc
Lead	
Butylins	
Butyltin ion	Tributyltin ion
Dibutyltin ion	
PAHs	
2-Methylnaphthalene	Benzo(k)fluoranthene
Acenaphthene	Chrysene
Acenaphthylene	Dibenzo(a,h)anthracene
Anthracene	Fluoranthene
Benzo(a)anthracene	Fluorene
Benzo(a)pyrene	Indeno(1,2,3-cd)pyrene
Benzo(b)fluoranthene	Naphthalene
Benzo(b+j)fluoranthene	Phenanthrene
Benzo(g,h,i)perylene	Pyrene
Phthalates	
BEHP	Diethyl phthalate
Butylbenzyl phthalate	Dimethyl phthalate
Dibutyl phthalate	Di-n-octyl phthalate
SVOCs	
1,4-Dichlorobenzene	Hexachlorobenzene
Benzoic acid	Hexachlorobutadiene
Carbazole	Isophorone
Phenols	
4-Chloro-3-methylphenol	Phenol
PCBs	
Aroclor 1242	Aroclor 1254
Aroclor 1248	Aroclor 1260
Total PCBs	
Dioxins and Furans	
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin	1,2,3,6,7,8-Hexachlorodibenzofuran
2,3,7,8-Tetrachlorodibenzofuran	1,2,3,7,8,9-Hexachlorodibenzofuran
1,2,3,7,8-Pentachlorodibenzo-*p*-dioxin	2,3,4,6,7,8-Hexachlorodibenzofuran
2,3,4,7,8-Pentachlorodibenzofuran	1,2,3,4,6,7,8-Heptachlorodibenzo-*p*-dioxin
1,2,3,7,8-Pentachlorodibenzofuran	1,2,3,4,6,7,8-Heptachlorodibenzofuran
1,2,3,4,7,8-Hexachlorodibenzo-*p*-dioxin	1,2,3,4,7,8,9-Heptachlorodibenzofuran

Table 5-1. Surface Water COIs for Fish

COIs	
1,2,3,6,7,8-Hexachlorodibenzo-*p*-dioxin	Octachlorodibenzo-*p*-dioxin
1,2,3,7,8,9-Hexachlorodibenzo-*p*-dioxin	Octachlorodibenzofuran
1,2,3,4,7,8-Hexachlorodibenzofuran	
Herbicides	
2,4-D	Dalapon
Pesticides	
2,4′-DDD	Endosulfan sulfate
2,4′-DDE	Endrin
2,4′-DDT	Endrin ketone
4,4′-DDD	Heptachlor
4,4′-DDE	Heptachlor epoxide
4,4′-DDT	alpha-HCH
Total DDTs	beta-HCH
Aldrin	delta-HCH
cis-Chlordane	gamma-HCH
trans-Chlordane	Methoxychlor
Total chlordane	cis-Nonachlor
Dieldrin	trans-Nonachlor
alpha-Endosulfan	Oxychlordane
beta-Endosulfan	
Perchlorate	
Perchlorate	

BEHP – bis(2-ethylhexyl) phthalate
COI – chemical of interest
HCH – hexachlorocyclohexane
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound

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Table 5-2. Summary of Surface Water Round 2 COPC Screen for Fish

Chemical	Unit	Maximium Surface Water Concentration	Chronic Eco SL	Round 2 COPC?[a]
Metals[b]				
Antimony (total)	mg/L	0.000125 T	0.03	no
Arsenic (dissolved)	mg/L	0.00064	0.15	no
Cadmium (dissolved)	mg/L	0.00003	0.00009	no
Chromium (dissolved)	mg/L	0.00033	0.0238	no
Copper (dissolved)	mg/L	0.00164 J	0.00274	no
Lead (dissolved)	mg/L	0.000152	0.00054	no
Nickel (dissolved)	mg/L	0.00112	0.0161	no
Selenium (total)	mg/L	0.0009 J	0.005	no
Silver (dissolved)	mg/L	0.000061	0.0001	no
Thallium (total)	mg/L	0.000032 JT	0.04	no
Zinc (dissolved)	mg/L	0.0419	0.0365	**yes**
Butylins				
Butyltin ion	µg/L	0.02 J	0.072	no
Dibutyltin ion	µg/L	0.0073 J	0.072	no
Tributyltin ion	µg/L	0.0023 J	0.072	no
PAHs				
2-Methylnaphthalene	µg/L	0.22	2.1	no
Acenaphthene	µg/L	0.21	23	no
Acenaphthylene	µg/L	0.043	306.9	no
Anthracene	µg/L	0.072	0.73	no
Benzo(a)anthracene	µg/L	0.11	0.027	**yes**
Benzo(a)pyrene	µg/L	0.15	0.014	**yes**
Benzo(b)fluoranthene	µg/L	0.11	0.6774	no
Benzo(b+j)fluoranthene	µg/L	0.006768	0.6415	no
Benzo(g,h,i)perylene	µg/L	0.14	0.4391	no
Benzo(k)fluoranthene	µg/L	0.1	0.6415	no
Chrysene	µg/L	0.19	2.042	no
Dibenzo(a,h)anthracene	µg/L	0.011 J	0.2825	no
Fluoranthene	µg/L	0.4	6.16	no
Fluorene	µg/L	0.063	3.9	no
Indeno(1,2,3-cd)pyrene	µg/L	0.11	0.275	no
Naphthalene	µg/L	0.77	12	no
Phenanthrene	µg/L	0.17	6.3	no
Pyrene	µg/L	0.45	10.11	no
Phthalates				
BEHP	µg/L	0.033	3	no
Butylbenzyl phthalate	µg/L	0.027 J	3	no
Dibutyl phthalate	µg/L	0.00598 J	3	no

Table 5-2. Summary of Surface Water Round 2 COPC Screen for Fish

Chemical	Unit	Maximium Surface Water Concentration	Chronic Eco SL	Round 2 COPC?[a]
Diethyl phthalate	µg/L	0.00665	3	no
Dimethyl phthalate	µg/L	0.00483	3	no
Di-n-octyl phthalate	µg/L	0.000142 J	3	no
SVOCs				
1,4-Dichlorobenzene	µg/L	0.019 J	15	no
Benzoic acid	µg/L	2.2 J	42	no
Carbazole	µg/L	0.06 J	18.6	no
Hexachlorobenzene	µg/L	0.00701	3.68	no
Hexachlorobutadiene	µg/L	0.00256	9.3	no
Isophorone	µg/L	0.018 J	130	no
Phenols				
4-Chloro-3-methylphenol	µg/L	0.65	0.6	**yes**
Phenol	µg/L	0.17 J	110	no
PCBs				
Aroclor 1242	µg/L	0.001083 NJ	0.053	no
Aroclor 1248	µg/L	0.00705 J	0.081	no
Aroclor 1254	µg/L	0.00657	0.033	no
Aroclor 1260	µg/L	0.013	94	no
Total PCBs	µg/L	0.018	0.014	**yes**
Dioxins and Furans				
2,3,7,8-TCDD[c]	pg/L	0.26	100	no
Herbicides				
2,4-D	µg/L	0.14 J	4	no
Dalapon	µg/L	0.23	2	no
Pesticides				
2,4′-DDD	µg/L	0.002057	0.001	**yes**
2,4′-DDE	µg/L	0.0000901	0.001	no
2,4′-DDT	µg/L	0.0187 NJ	0.001	**yes**
4,4′-DDD	µg/L	0.00325	0.001	**yes**
4,4′-DDE	µg/L	0.000663	0.001	no
4,4′-DDT	µg/L	0.003862	0.001	**yes**
Total DDTs	µg/L	0.0199 NJ	0.001	**yes**
Aldrin	µg/L	0.00001627 J	0.3	no
cis-Chlordane	µg/L	0.00104 J	0.0043	no
trans-Chlordane	µg/L	0.00108 J	0.0043	no
Total chlordane	µg/L	0.00293 J	0.0043	no
Dieldrin	µg/L	0.0000625	0.056	no
alpha-Endosulfan	µg/L	0.000569	0.056	no
beta-Endosulfan	µg/L	0.0000357	0.056	no
Endosulfan sulfate	µg/L	0.00077 J	0.056	no

Table 5-2. Summary of Surface Water Round 2 COPC Screen for Fish

Chemical	Unit	Maximium Surface Water Concentration	Chronic Eco SL	Round 2 COPC?[a]
Endrin	µg/L	0.00000157 J	0.036	no
Endrin ketone	µg/L	0.00000233 J	0.036	no
Heptachlor	µg/L	0.00000788 J	0.0038	no
Heptachlor epoxide	µg/L	0.000005262 J	0.0038	no
alpha-HCH	µg/L	0.00009525 J	0.08	no
beta-HCH	µg/L	0.00003474 J	0.08	no
gamma-HCH	µg/L	0.000768	0.08	no
delta-HCH	µg/L	0.00167 NJ	0.08	no
Methoxychlor	µg/L	0.0111	0.03	no
cis-Nonachlor	µg/L	0.00001577 J	0.0043	no
trans-Nonachlor	µg/L	0.00004879 J	0.0043	no
Oxychlordane	µg/L	0.00000267 J	0.0043	no
Perchlorate				
Perchlorate	µg/L	15.7 T	18	no

[a] Round 2 COPCs were identified where the maximum detected surface water concentration was greater than the chronic Eco SL.

[b] Concentrations of metals were evaluated as total or dissolved concentrations, depending on the Eco SL.

[c] 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin was evaluated as a representative for all dioxins and furans.

COPC – chemical of potential concern
BEHP – bis(2-ethylhexyl) phthalate
Eco SL – Ecological Screening Level
J – estimated concentration
N – presumptive evidence of a compound
HCH – hexachlorocyclohexane
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
T – value calculated or selected from multiple results
TCDD – tetrachlorodibenzo-*p*-dioxin

Table 5-3. Round 2 COPCs Identified in TZW for Fish

Round 2 Chemicals of Potential Concern	
Metals	
Barium (total)[a]	Nickel (dissolved)[b]
Cadmium (dissolved)[b]	Sodium (total)[a]
Copper (dissolved)[b]	Vanadium (total)[a]
Lead (dissolved)[b]	Zinc (dissolved)[b]
PAHs	
2-Methylnaphthalene	Chrysene
Acenaphthene	Dibenzo(a,h)anthracene
Anthracene	Fluoranthene
Benzo(a)anthracene	Fluorene
Benzo(a)pyrene	Indeno(1,2,3-cd)pyrene
Benzo(b)fluoranthene	Naphthalene
Benzo(g,h,i)perylene	Phenanthrene
Benzo(k)fluoranthene	Pyrene
SVOCs	
1,2-Dichlorobenzene	Dibenzofuran
1,4-Dichlorobenzene	
Pesticides	
2,4′-DDD	4,4′-DDE
2,4′-DDT	4,4′-DDT
4,4′-DDD	Total DDTs
Herbicides	
Dalapon	Silvex™
VOCs	
1,1-Dichloroethene	Ethylbenzene
1,2,4-Trimethylbenzene	Isopropylbenzene
1,3,5-Trimethylbenzene	Toluene
Benzene	Trichloroethene
Carbon disulfide	Vinyl chloride
Chlorobenzene	m,p-Xylene
Chloroethane	o-Xylene
cis-1,2-Dichloroethene	Total xylenes
Cyanide	
Cyanide	
Perchlorate	
Perchlorate	

[a] Eco SL is based on a total concentration; dissolved concentrations were not evaluated.
[b] Eco SL is based on a dissolved concentration; total concentrations were not evaluated.

COPC – chemical of potential concern
TWZ – transition zone water
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolitile organic compound
VOC – volatile organic compound

Table 5-4. Studies Evaluated for Effects of Copper on Avoidance Behavior in Fish

Species	Copper Concentration Effect Level (µg/L)[a]	Water Hardness (mg Calcium Carbonate/L)	Source
Rainbow trout	0.1	89.5	Folmar (1976) as cited in Brooks (2004)
Chinook salmon	0.8[b]	25.3	Hansen et al. (1999b)
Rainbow trout	1.6 – 88[c]	25.3	Hansen et al. (1999b)
Atlantic salmon	2.3	20	Sprague et al. (1964)
Rainbow trout	4.4	23.0 – 27.0	Giattina et al.(1982)
Juvenile coho	4.4[d]	120	Sandahl et al. (2004)
Coho salmon	< 6.4	30.5	Rhenberg and Schreck (1986) as cited in Brooks (2004)
Rainbow trout	8	90	Hara et al (1976) as cited in Brooks (2004)
Juvenile coho	13[e]	100	(Baldwin et al. 2003)
Rainbow trout	< 22[f]	61.8 – 64.0	Saucier et al (1991) as cited in Brooks (2004)
Brown trout	55	157.8	Baldigo and Baudanza (2001) as cited in Brooks (2004)
Rainbow trout	70	112.4	Black and Birge (1980) as cited in Brooks (2004)

[a] Effect level is the lowest reported concentration where avoidance behavior was observed for a given species in each toxicological study reviewed.

[b] Chinook salmon avoided copper levels up to 22.5 µg/L but failed to avoid levels above the acutely lethal concentration of 44 µg/L (Hansen et al. 1999b).

[c] Rainbow trout avoided copper levels up to 88 µg/L but failed to avoid above the acutely lethal concentration of 180 µg/L (Hansen et al. 1999b).

[d] Fish demonstrated a 20% loss of olfactory function following a 7-day exposure at 4.4 µg /L. Fish exposed to 11.1 µg/L experienced a 50% loss of olfactory function, and fish exposed to 20 µg/L experienced a 90% loss of olfactory function (Sandahl et al. 2004).

[e] Fish experienced a 50% loss of olfactory function when exposed to 13 µg/L copper for 30 minutes (Baldwin et al. 2003).

[f] Copper concentrations were measured as Cu^{2+}.

APPENDIX G: ECOLOGICAL RISK ASSESSMENT

Attachment G5
Round 2 Toxicity Reference Value Addendum for Fish and Wildlife

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LIST OF ACRONYMS

bw/day	body weight per day
COI	chemical of interest
EPA	US Environmental Protection Agency
ERA	ecological risk assessment
dw	dry weight
IRIS	Integrated Risk Information System
LOAEL	lowest-observed-adverse-effect level
NOAEL	no-observed-adverse-effect level
NTP	National Toxicology Program
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PRE	preliminary risk evaluation
RI/FS	remedial investigation/feasibility study
TBT	tributyltin
TRV	toxicity reference value
ww	wet weight

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1.0 INTRODUCTION

The following is the Round 2 toxicity reference value (TRV) addendum to the TRV technical memorandum submitted as Appendix B to the Ecological Preliminary Risk Evaluation (Windward 2005), hereafter referred to as the Ecological PRE. All TRVs recommended in the TRV technical memorandum were used in the Round 2 Ecological Risk Assessment (ERA), with the exception of those presented in this addendum. Changes to selected TRVs for the Round 2 ERA were based on the following:

- None of the selected Round 2 TRVs are based on studies that measured effects in field-collected organisms, had a value based on behavior endpoints, or reported tissue concentrations that were measured in individuals that were poorly representative of those in which adverse effects were observed. Behavior studies were not considered direct measures of the survival, growth, and mortality endpoints that were selected and agreed upon for this ERA. Toxicity studies based on field-collected organisms are not appropriate for evaluating risks beyond a screening-level evaluation because the presence of multiple chemicals and other environmental factors may result in adverse effects that complicate the interpretation of field study results. Studies that reported tissue concentrations that were measured in poorly representative individuals do not represent critical tissue thresholds. While these studies were acceptable for the development of screening level TRVs using the Ecological PRE and used to identify chemicals of potential concern, these studies do not represent appropriate thresholds for the Round 2 risk evaluation and subsequent risk management decisions.
- Changes to assumptions made in dietary dose TRVs changed the value of selected TRV or resulted in lower TRVs than those selected in Appendix B of the Ecological PRE.
- Additional review of toxicological studies since the preparation of Appendix B of the Ecological PRE resulted in lower TRVs than those selected in Appendix B of the Ecological PRE.

The following sections presents the additional toxicological studies reviewed since Appendix B of the Ecological PRE, changes to assumptions used to derive dietary dose TRVs, and any changes in selected TRVs for the Round 2 ERA. Changes to fish and wildlife TRVs for the Round 2 ERA are presented in Table 1. Rational for changes in the TRV selection process for fish and results of revised assumptions for calculating dietary dose TRVs for fish are summarized in Section 2.0. Rationale for the changes to the TRV selection process for birds and mammals, and the results of additional toxicological literature reviewed are presented in Section 3.0.

2.0 ROUND 2 FISH TRVS

Selected fish tissue residue and dietary dose TRVs were updated from Appendix B of the Ecological PRE for some chemicals, based on additional toxicological studies identified and reviewed, and based on the Round 2 ERA TRV selection criteria. These TRVs were used in the Round 2 ERA. Updated selected fish tissue residue and dietary dose TRVs for the Round 2 ERA are summarized in the following subsections.

2.1 FISH TISSUE RESIDUE TRVS

Selected whole-body tissue residue TRVs for polychlorinated biphenyls (PCBs) and mercury were updated for the Round 2 ERA from Appendix B of the Ecological PRE. Changes in the TRV selection process were made for these two chemicals so that none of the selected Round 2 TRVs were based on studies that measured effects in field-collected organisms, had a value based on behavior endpoints, or reported tissue concentrations that were measured in individuals poorly representative of those in which adverse effects were observed.

2.1.1 PCBs

Appendix B of the Ecological PRE presents the details on the twenty-nine papers on the potential adverse effects of PCB mixtures on fish that were reviewed for TRV selection. Whole-body tissue residue[1] lowest-observed-adverse-effect levels (LOAELs) based on PCB Aroclor mixtures ranged from 1.53 mg/kg ww for mortality of field-collected brook trout fry to 645 mg/kg ww for growth and mortality of fingerling coho salmon (Mayer et al. 1977). A lower LOAEL of 0.15 mg/kg based on injected dose was reported in Folmar et al. (1982) in which yearling coho salmon survival was reduced; however, no tissue residue concentrations were measured in the study. Additionally, Hugla and Thome (1999), reported a LOAEL of 0.52 mg/kg ww (converted from 2.6 mg/kg dw assuming 80% moisture content) for reduced fecundity in common barbel, however, this concentration was measured 1 year prior to when effects were observed and based on the number of fish included in various analyses, it appears that this concentration was measured in male fish, and did not include liver tissue. In a separate study, Hugla et al. (1995) report that hatchery male common barbel were about two times smaller than females and that PCB body burdens increase with fish size. Therefore, male PCB body burdens appear to be poorly representative of those in females. The experimental design for Hugla and Thome (1999) was poorly reported but it appears that there was no replication of exposures and the statistics are based on pseudoreplication. At the LOAEL concentration for the fecundity endpoint, Hugla and Thome (1999) reported no adverse effects on egg weight, hatching rate, or spawning success in the first reproductive season. No other common barbel PCB

[1] Whole-body tissue concentrations were either as reported in the paper or were modeled from egg concentration data.

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toxicity data are available to evaluate this species sensitivity relative to other fish. Because of the uncertainty in the reported tissue concentrations in this study, it was not selected for TRV derivation.

At the lowest LOAEL, Berlin et al. (1981) exposed field-collected eggs to various levels of DDE concentrations via diet and water for 176 days, and fry mortality was observed at all exposure levels. However, the field-collected eggs contained 7.6 mg/kg of PCB and 4.7 mg/kg of DDE in the tissue and possibly other, uncharacterized organic chemicals that could have contributed to the reported toxicity. Additionally, not all fish present at the start of the experiment could be accounted for at the end of the experiment.

Van Wezel et al. (1995) reported a range of lethal tissue concentrations from 1.86 to 749 mg/kg ww in 6- and 12-month-old fathead minnows exposed to Aroclor 1242, 1254, or 1260 for up to 12 days. Lethal tissue concentrations in individual fish that died in less than 20 hours ranged from 1.86 to 30 mg/kg ww, whereas lethal tissue concentrations in individual fish that died in 100 to 300 hours were 120 to 749 mg/kg ww. The authors hypothesized that whole-body concentrations in fish that died within 30 hours did not reach equilibrium with concentrations at the site of toxic action. However, this study did not include a control, and 50% variability in lethal tissue concentrations was reported. This study also did not report tissue concentrations associated with a given exposure level. Based on the variability in the data and lack of dose-response information, no LOAEL or lowest-observed-adverse-effect level (NOAEL) could be reliably determined from this paper.

At the next lowest LOAEL, Fisher et al. (1994) exposed Atlantic salmon eyed embryos to 62.5 to 62,500 µg/L PCB concentrations in water for 48 hours, and fry growth was reduced at all exposures. The lowest egg concentration was 0.857 mg/kg ww, which was converted into an adult tissue concentration of 4.02 mg/kg ww using the adult: fertilized egg conversion factor of 4.69. This was the lowest LOAEL reported in the literature reviewed and was selected as the LOAEL TRV because it represents the most conservative LOAEL value. Use of this LOAEL is uncertain, however, because of uncertainty associated with application of the egg: adult conversion factor. As indicated by several toxicological studies, fish embryos appear to be the most sensitive life stage to PCB exposure. However, PCB concentrations in egg tissues are not available from the study area for comparison to egg-tissue TRVs, and the use of an extrapolation factor allows for the protection of a sensitive life stage by comparing available adult whole-body tissues from the study area to an estimated maternal adult tissue concentration. Use of the conversion factor may over- or underestimate whole-body tissue concentrations in an adult, which could result in adverse effects on growth of offspring. In addition to the use of an extrapolation factor, there is uncertainty associated with the representativeness of the exposure conditions used by Fisher et al. (1994) to the exposure conditions in the study area. Fisher et al. exposed alevins to high concentrations of PCBs in water (625 µg/L) for a short exposure duration (48 hours). While early stage development is considered

representative of a critical life stage (and chronic conditions), exposure to high levels of PCBs over a short period is not representative to the exposure conditions of the fish receptors that spawn in the study area.

NOAELs ranged from 0.98 mg/kg ww, where no effect on growth or survival was reported in juvenile chinook salmon (Powell et al. 2003) to 120 mg/kg ww for no effect on survival of juvenile rainbow trout (Mayer et al. 1985). The highest reproduction NOAEL found in the literature reviewed that was lower than the selected LOAEL was based on Hansen et al (1973) where fry survival was unaffected in sheepshead minnow offspring following maternal exposure for 28 days. This NOAEL (1.9 mg/kg ww) was selected as the NOAEL TRV because it was the highest NOAEL reported below the selected LOAEL TRV and was based on a reproductive endpoint.

2.1.2 Mercury

Appendix B of the Ecological PRE (Windward 2005) presents the details on the twenty toxicity studies that measured effects and concentrations associated with mercury exposure and that were evaluated for TRV selection. Whole-body tissue residue[2] LOAELs ranged from 0.23 mg/kg ww for predator avoidance behavior of golden shiners (Webber and Haines 2003) to 1,045 mg/kg ww for hatchability of Japanese medaka (Heisinger and Green 1975). At the lowest LOAEL, Webber and Haines (2003) reported increased shoal area following exposure to a model predator compared to controls. At this concentration no effects were observed for two other predation avoidance behaviors, shoal height, and time to settling. The authors state that these behaviors may increase the chance of predation but note that this has not been confirmed in wild fish. Because these behavioral effects cannot be tied directly to effects on growth mortality or reproduction, this study was not selected for the Round 2 ERA.

At the next lowest LOAEL of 0.254 mg/kg ww, Friedmann et al. (1996) reported reduction in growth for walleye; however, this LOAEL was based on the whole-body tissue concentrations minus viscera concentrations. In addition, the observed growth effect was not dose-responsive, and mortality was elevated in the controls. Therefore, this LOAEL was not selected for the Round 2 ERA.

At the next lowest LOAEL of 0.47 mg/kg ww, wild caught adult male mummichog exposed to dietary methyl-mercury in for 42 days experienced increased mortality relative to controls. Mortality did not follow a typical dose-response pattern; in that fish with tissue residues of 1 and 11 µg/g had similar or lower mortality than those with tissue residues of 0.5 µg/g. When male and female mortality were combined, there was no significant difference between mercury-dosed fish and controls. The LOAEL of 0.47 mg/kg ww reported in Matta et al. (2001) was selected as the tissue

[2] Whole-body tissue concentrations were either as reported in the paper or were modeled from egg concentration data. See Appendix B of the Ecological PRE for details on egg-to-adult tissue concentration conversion factors.

LOAEL TRV for mercury in fish. The NOAEL of 0.2 for mortality of male mummichog from the same study was also selected. This was the lowest NOAEL reported in the reviewed studies.

The selected LOAEL was the lowest whole-body tissue residue for growth, mortality, or reproduction reported in the literature reviewed. This LOAEL is at the lower limit of the 0.5 to 1.0 mg/kg ww range Beckvar et al. (1996) cite as concentrations where reproductive and early life stage effects possibly start to occur for fish.

2.2 FISH DIETARY DOSE TRVS

The assumptions used to derive dietary dose-based TRVs for fish presented in Appendix B of the Ecological PRE (Windward 2005) were revisited for the Round 2 ERA. Fish ingestion rates were evaluated in greater detail, specifically for those studies in which neither a dietary dose nor a feeding rate was presented in a given toxicological study. For such studies, the dose was estimated using the average feeding rate from toxicological studies for the same species of fish fed a similar food type (i.e., live prey or a synthetic diet).

Fish dietary TRVs are presented as a daily dose, expressed as mg/kg bw/day. Most studies reported toxicity results as the chemical concentration in food associated with adverse effects, although a few studies presented results as a daily dose. The following approach was used to determine appropriate values for body weight, percent moisture, and food ingestion rate for use in the dose calculation. All body weight, food ingestion rate, and percent moisture values used in the calculation of dietary dose fish TRVs are presented in Table 2.

Body Weight —Body weight data from the study were used if reported. If body weight data were presented for different times during the exposure period, these data were used to generate an average body weight (i.e., where average body weight [BW] = $BW_{initial}$ + ½ BW gain over duration of exposure) following the convention used in the toxicity literature (e.g., Cockell et al. 1991). If final body weights were not presented, food ingestion rates and IR_{diet} were calculated based on the initial body weight. If body weight data were presented for each dietary concentration level, average body weights were calculated by concentration level, and these weights were used to derive corresponding NOAEL and LOAEL daily dietary dose TRVs. If no body weight data were provided in the study or data provided were not considered representative, body weights were estimated from other literature sources or toxicity studies.

Food Ingestion Rate—Ingestion rate data from the study for the corresponding NOAEL or LOAEL doses were applied, if provided. If only an average for all individuals was provided, this was assumed to be representative and applied. If no ingestion rates were provided in the study, they were estimated from other literature sources for the same species, or if no literature was available, an ingestion rate of 2% food (dw)/kg bw/day was assumed as a conservative estimate based on the food ingestion rates commonly reported for laboratory toxicity studies.

Percent Moisture—Moisture content of food and the weight basis of the feeding rate is frequently not reported in dietary toxicity studies. The weight basis of ingestion rate and dietary concentration are immaterial to calculation of daily dietary doses, so long as the weight basis is the same. Food concentrations were converted, as necessary, to the corresponding basis of the food ingestion rate, (ww or dw) using percent moisture in food. If the weight basis of the dietary concentration was not reported, commercial feed or pelleted diet was assumed to approximate a dw concentration, and eighty percent moisture was assumed when the diet consisted of organism prey (e.g., invertebrate prey).

Details on the assumptions and sources of dietary feeding rates and the derivation of dietary dose TRVs are presented in the following subsections and in Table 2. Selected dietary dose TRVs for fish for the Round 2 ERA are also presented.

2.2.1 Metals

Arsenic

Six toxicity studies that measured the effects of dietary arsenic on fish were evaluated for TRV selection (Blazer et al. 1997; Cockell and Bettger 1993; Cockell and Hilton 1988; Cockell et al. 1991, 1992; Oladimeji et al. 1984). Studies were conducted with two species of fish (rainbow trout and striped bass) all evaluating the growth endpoint. Hockett et al. (2003) was reviewed; however, it was not included in the Round 2 ERA TRV selection process because it exists only as an abstract and poster from a conference proceeding and has not been published as a manuscript.

Table 2 summarizes the feeding rate assumptions, dietary dose calculations, and dietary NOAELs and LOAELs for arsenic reported in the reviewed literature. All of the six studies that measured the effects of dietary arsenic on fish reported either an exposure rate or a feeding rate from which an exposure rate could be calculated. Dietary dose LOAELs ranged from 0.50 mg/kg bw/day for growth of juvenile rainbow trout (Cockell et al. 1991) to 2.9 mg/kg bw/day for growth of rainbow trout (Cockell and Hilton 1988). Dietary NOAELs ranged from 0.12 mg/kg bw/day for growth of rainbow trout ((Cockell et al. 1991) to 0.55 mg/kg bw/day for growth of rainbow trout (Cockell et al. 1992). At the lowest LOAEL (0.50 mg/kg bw/day), adverse effects on growth were attributed to reduced feeding relative to controls (Cockell et al. 1991). At the next lowest LOAEL, Oladimeji et al. (1984) reported lower weight gain than control fish for rainbow trout exposed to 0.60 mg/kg bw/day arsenic (as sodium

arsenite) for 2, 4, or 6 weeks. No effects were observed at the next lower dose of 0.40 mg/kg bw/day. The LOAEL and NOAEL, 0.60 and 0.40 mg/kg bw/day, respectively reported in (Oladimeji et al. 1984) were selected as the dietary dose TRVs because these were the lowest doses reported where effects were attributed to arsenic toxicity.

Cadmium

Seven studies measuring the effects of dietary cadmium on fish were evaluated for TRV selection (Baldisserotto et al. 2005; Franklin et al. 2005; Hatakeyama and Yasuno 1982, 1987; Mount et al. 1994; Handy 1993b; Kang et al. 2005; Kim et al. 2004). Erickson et al. (2003) was reviewed; however, it was not included in the Round 2 ERA TRV selection process because it exists only as an abstract and poster from a conference proceeding and has not been published as a manuscript.

Table 2 summarizes the feeding rate assumptions, dietary dose calculations, and dietary NOAELs and LOAELs for cadmium reported in the reviewed literature. Dietary dose NOAELs or LOAELs were reported for four species (Atlantic salmon, guppy, rainbow trout and rockfish). Adverse effects in fish at various life stages were reported (i.e., fry, juvenile, and adult); effects included reduced survival, reduced growth, reduced fry production, and reduced fry survival. Six of the seven studies that measured the effects of dietary cadmium on fish reported either an exposure rate or a feeding rate from which an exposure rate could be calculated (Hatakeyama and Yasuno 1987; Handy 1993b; Mount et al. 1994; Kim et al. 2004; Kang et al. 2005; Franklin et al. 2005; Baldisserotto et al. 2005). Insufficient data were presented in Hatekeyama and Yasuno (1982) to calculate an exposure rate or feeding rate, the average feeding rate presented in Hatekeyama and Yasuno (1987) with the same species was assumed.

Dietary dose LOAELs ranged from 0.01 mg/kg bw/day for growth of juvenile rockfish (Kang et al. 2005; Kim et al. 2004) to 68 mg/kg bw/day for mortality of rainbow trout (Handy 1993b). Dietary NOAELs ranged from 2.5 mg/kg bw/day for mortality of juvenile rockfish (Kang et al. 2005; Kim et al. 2004) to 29 mg/kg bw/day for reproduction of guppy (Hatakeyama and Yasuno 1987). The lowest dietary dose LOAEL of 0.01 mg/kg bw/day, for growth of juvenile rockfish was selected as the LOAEL dietary dose TRV for cadmium (Kang et al. 2005; Kim et al. 2004). Kang et al. (2005) and Kim et al. (2004), exposed juvenile rockfish to 0.5, 5, 25, or 125 mg/kg dw of cadmium as cadmium nitrate for 60 days. Significant effects on growth (identified as condition factor, body weight growth rate, and body length growth rate) were reported for fish exposed to all four dietary concentrations. The lowest concentration was administered at a dose of 0.01 mg/kg bw/day. Toxic effects are somewhat uncertain because in one of the two papers where the study is reported, (Kim et al. 2004), the observed growth effect is partially attributed to reduced food intake which may be due to food avoidance rather than toxicological effects. The lowest LOAEL was two to three orders of magnitude lower than the NOAELs reported in the six other studies (2.5 to 29 mg/kg bw/day) and was three orders of magnitude lower than the LOAELs reported in the three other studies that reported

LOAELs (35 to 68 mg/kg bw/day). As such, although there were few species evaluated, the dose based LOAEL TRV is likely conservative because the majority of the toxicological studies reviewed indicate that the selected LOAEL may over predict cadmium toxicity (by several orders of magnitude). No NOAEL was available that was lower than the selected LOAEL TRV, so none was selected.

Chromium

No relevant literature addressing the toxicity of dietary chromium to fish was found. One study on the toxicity of chromium to fish was reviewed (Walsh et al. 1994), however, the fish tested (gray mullet) were exposed to chromium in field-simulated conditions in which sediment, algae, and clams were treated with chromium and then placed in an aquarium. No effect on growth or survival of gray mullet was reported following exposure, and no dietary concentration could be derived from this study. Consequently, no NOAEL or LOAEL was selected for chromium.

Copper

Thirteen toxicity studies that exposed fish to dietary copper were evaluated for TRV selection (Berntssen et al. 1999a, b; Baker et al. 1998; Handy 1992, 1993b; Kamunde et al. 2001; Kang et al. 2005; Lanno et al. 1985a, b; Lundebye et al. 1999; Miller et al. 1993; Mount et al. 1994; Murai et al. 1981). Adverse effects on growth or survival were reported in four species (Atlantic salmon, channel catfish, rainbow trout, and grey mullet) following exposure to dietary copper. Erickson et al. (2003) was reviewed; however, it was not included in the Round 2 ERA TRV selection process because it exists only as an abstract and poster from a conference proceeding and has not been published as a manuscript.

Table 2 summarizes the feeding rate assumptions, dietary dose calculations, and dietary NOAELs and LOAELs for copper reported in the reviewed literature. Ten of the thirteen studies that measured the effects of dietary copper on fish reported either an exposure rate or a feeding rate from which an exposure rate could be calculated (Baker et al. 1998; Berntssen et al. 1999a, b; Handy 1993b; Kamunde et al. 2001; Lanno et al. 1985a, b; Miller et al. 1993; Mount et al. 1994; Handy 1992). Feeding rate data for Murai et al. (1981) were reported in a separate study (Murai and Andrews 1978). Two studies did not report sufficient data to estimate an exposure rate (Lundebye et al. 1999; Miller et al. 1993). A feeding rate of 2.8% bw/day was assumed for Lundebye et al. (1999) based on the average feeding rate for Atlantic salmon fed synthetic diets in 2 toxicity studies (Berntssen et al. 1999a, b). A feeding rate of 1.9% bw/day was assumed for Miller et al. (1993) based on the average feeding rate of rainbow trout fed a synthetic diet in 16 dietary toxicity studies (Baldisserotto et al. 2005; Cockell and Bettger 1993; Cockell and Hilton 1988; Cockell et al. 1991, 1992; Franklin et al. 2005; Galvez and Wood 1999; Handy 1993a, b; Hendricks et al. 1985; Lanno et al. 1985a, b; Kamunde et al. 2001; Macek et al. 1970; Oladimeji et al. 1984; Rodgers and Beamish 1982).

Dietary dose LOAELs ranged from 0.48 mg/kg bw/day for growth of channel catfish fed dietary copper for 16 weeks (Murai et al. 1981) to 60 mg/kg bw/day for growth of Atlantic salmon fry (Baker et al. 1998). Dietary NOAELs ranged from 0.24 mg/kg bw/day for growth of channel catfish (Murai et al. 1981) to 69 mg/kg bw/day for mortality of rainbow trout (Handy 1993b). Erickson et al. (2003) report a dietary concentration NOAEL for channel catfish growth following exposure for 30 days to live copper contaminated prey. However, this study did not report an exposure rate or feeding rate, so the dietary dose is uncertain. Assuming a feeding rate of 6% body weight per day results in a dietary dose NOAEL of 14 mg/kg bw/day which is much higher than that of 0.24 mg/kg bw/day from Murai et al. (1981). The dietary concentration NOAEL of 246 mg/kg dw reported by Ericson et al. (2003) was also much higher the 18mg/kg dw LOAEL concentration reported in Murai et al. (1981). Because Erickson et al. exposed fish to live prey, the exposures were more ecologically realistic and the resulting growth NOAEL suggests that channel catfish sensitivity to copper may be much lower than reported by Murai et al. (1981). Ericson et al. (2003) has not been published in manuscript form so it was not included in the studies evaluated for TRV derivation. Therefore, the lowest LOAEL, 0.48 mg/kg bw/day for reduced growth of channel catfish, was selected as the LOAEL TRV because it was the most conservative LOAEL reported in the reviewed literature. The lowest NOAEL of 0.24 mg/kg bw/day reported by Murai et al. (1981) was selected as the dietary NOAEL TRV, where no effect on weight gain was observed.

Lead

Two studies that measured the toxicity of dietary lead in fish were evaluated for TRV selection (Goettl et al. 1976; Mount et al. 1994) Mount et al. 1994). Adverse effects on growth or survival were reported in one species (rainbow trout) following exposure to dietary lead. Erickson et al. (2003) was reviewed; however, it was not included in the Round 2 ERA TRV selection process because it exists only as an abstract and poster from a conference proceeding and has not been published as a manuscript.

Table 2 summarizes the feeding rate assumptions, dietary dose calculations, and dietary NOAELs and LOAELs for zinc reported in the reviewed literature. One of the two studies that measured the effects of dietary lead to fish reported a feeding rate from which an exposure rate could be directly calculated (Mount et al. 1994); the other study did not report sufficient data to estimate a feeding rate or exposure rate (Goettl et al. 1976) so a feeding rate of 1.9% bw/day was assumed for based on the average feeding rate of rainbow trout fed a synthetic diet in 16 dietary toxicity studies (Baldisserotto et al. 2005; Cockell and Bettger 1993; Cockell and Hilton 1988; Cockell et al. 1991, 1992; Franklin et al. 2005; Galvez and Wood 1999; Handy 1993a, b; Hendricks et al. 1985; Lanno et al. 1985a, b; Kamunde et al. 2001; Macek et al. 1970; Oladimeji et al. 1984; Rodgers and Beamish 1982).

No LOAELs were reported. The highest dietary dose NOAEL is 134 mg/kg bw/day for no effect on rainbow trout mortality and growth following exposure to lead in a

synthetic diet for 191 days (Goettl et al. 1976). This NOAEL is based on an estimated feeding rate and is unbounded by a LOAEL and is therefore, highly uncertain.

Selenium

Three toxicity studies that measured the effects of dietary selenium on fish were evaluated for TRV selection (Cleveland et al. 1993; Coughlan and Velte 1989; Ogle and Knight 1989). In the three studies evaluated for TRV selection, mortality or reduced growth was reported in juvenile bluegill, striped bass, and fathead minnow. Table 2 summarizes all dietary fish TRVs for selenium reported in the reviewed literature including the feeding rate assumptions, dietary dose calculations, and dietary NOAELs and LOAELs for selenium reported in the reviewed literature. All of the three studies that measured the effects of dietary selenium on fish reported either a feeding rate or an exposure rate.

Dietary dose LOAELs ranged from 0.2 mg/kg bw/day for mortality of bluegill (Cleveland et al. 1993) to 1.2 mg/kg bw/day for growth of fathead minnow (Ogle and Knight 1989). At the lowest LOAEL, Cleveland et al. (1993) reported mortality of juvenile bluegill fed a synthetic diet enriched with 6.6 µg/g ww seleno-L-methionine for 90 days. NOAELs ranged from 0.1 mg/kg bw/day for mortality of bluegill (Cleveland et al. 1993) to 1.8 mg/kg bw/day for reproduction of fathead minnow (Ogle and Knight 1989). The lowest LOAEL and lowest NOAEL of 0.10 and 0.20 mg/kg bw/day, respectively, reported in Cleveland et al. (1993) were selected as NOAEL and LOAEL TRVs. There is uncertainty associated with the selected TRVs because few studies were evaluated.

Silver

One study that measured the toxicity of dietary silver to fish was evaluated for TRV selection (Galvez and Wood 1999). This study exposed rainbow trout to dietary silver and observed no adverse effect on growth. Galvez and Wood (1999) reported a feeding rate, so the dose could be directly calculated. Table 2 summarizes the feeding rate assumptions, dietary dose calculations, and dietary NOAEL for silver reported in Galvez and Wood (1999). The NOAEL reported in this study, 70 mg/kg bw/day, was selected as the NOAEL TRV. No LOAEL was selected for silver because none was identified in the literature. There is high uncertainty associated with the selected NOAEL because the literature dataset for the dietary toxicity of silver to fish is limited to one study reporting an unbounded NOAEL.

Zinc

Two fish studies that measured the toxicity of dietary zinc were evaluated for TRV selection (Mount et al. 1994; Takeda and Shimma 1977). Effects on growth and mortality were reported in all studies following exposure to dietary zinc. Erickson et al. (2003) was reviewed; however, it was not included in the Round 2 ERA TRV selection process because it exists only as an abstract and poster from a conference proceeding and has not been published as a manuscript.

Table 2 summarizes the feeding rate assumptions, dietary dose calculations, and dietary NOAELs and LOAELs for zinc reported in the reviewed literature. Only one of the two studies that measured the effects of dietary zinc on fish reported a feeding rate from which an exposure rate could be directly calculated (Mount et al. 1994). Takeda and Shimma (1977) did not report a feeding rate or exposure rate so a feeding rate of 1.9% bw/day was assumed based on the average feeding rate of rainbow trout fed a treated diet in 16 dietary toxicity studies (Baldisserotto et al. 2005; Cockell and Bettger 1993; Cockell and Hilton 1988; Cockell et al. 1991, 1992; Franklin et al. 2005; Galvez and Wood 1999; Handy 1993a, b; Hendricks et al. 1985; Lanno et al. 1985a, b; Kamunde et al. 2001; Macek et al. 1970; Oladimeji et al. 1984; Rodgers and Beamish 1982).

Only one LOAEL was reported in the two studies reviewed. Takeda and Shimma (1977) reported a reduction in growth in fingerling rainbow trout following six weeks of dietary dose of 40 mg/kg bw/day and this dietary dose was selected as the LOAEL TRV. The NOAEL TRV of 19 mg/kg bw/day from the same study was selected as the NOAEL TRV. There is high uncertainty associated with the selected TRVs because the literature dataset for the dietary toxicity of zinc to fish is limited to two studies. These TRVs are also uncertain because they are based on an assumed feeding rate of 1.9% bw/day.

Tributyltin

Only one study that measured the toxicity of dietary tributyltin (TBT) to fish (Shimasaki et al. 2003) was reviewed. Table 2 summarizes the dietary dose calculations, feeding rate assumptions, dietary dose calculations, and dietary NOAELs and LOAELs from this study. Shimasaki et al. did not report an exposure rate or feeding rate and no other dietary toxicity studies in which winter flounder, or closely related fish, were fed a synthetic diet were identified so a feeding rate of 2% bw/day was assumed.

Shimasaki et al. (2003) reported the effects on reproductive success of Japanese flounder larvae following exposure to dietary TBT. The flounder larvae that were exposed to TBT were generated from two parents that were genetically female (with XX chromosomes); however, one parent was phenotypically and functionally male, and offspring were generated. All offspring produced were genetically female (XX). The female offspring larvae were exposed to TBT as tributyltin oxide for approximately 65 days at two different doses, and the effects on growth and sex reversal were measured relative to a control. A significant reduction in growth was reported at 100 days at both 0.0021 mg/kg bw/day and 0.020 mg/kg bw/day treatments; however, at the lower dose, growth was recovered by 300 days so this was not considered an adverse effect. An increase in sex reversal was reported in TBT-exposed fish. The actual impact of TBT on sex reversal is unclear because sex reversal was apparent in the parent fish, prior to TBT exposure; and sex reversal did not appear to affect production of offspring. Because the ecological significance of the sex reversal endpoint is uncertain, the LOAEL of 0.020 mg/kg bw/day associated with the

growth endpoint was selected as the dietary LOAEL TRV for TBT. The growth NOAEL of 0.0021 mg/kg bw/day was selected as the NOAEL TRV. No difference in the concentration was calculated converting the LOAEL TRV for TBT from tributyltin oxide into tributyltin ion (due to significant figures). There is additional uncertainty associated with the selected TRVs because the literature dataset for the dietary toxicity of TBT to fish is limited to one study. There is high uncertainty associated with the selected TRVs because the literature dataset for the dietary toxicity of TBT to fish is limited to one study. These TRVs are also uncertain because they are based on an assumed feeding rate of 2% bw/day.

2.2.2 Total and Individual PAHs

Adverse effects in fish resulting from polycyclic aromatic hydrocarbon (PAH) exposure were considered using for benzo(a)pyrene and total PAHs. Five toxicity studies measuring the effects of dietary PAHs to fish were evaluated for TRV selection. Four of the studies reviewed evaluated the toxicity of benzo(a)pyrene to fish (Hart and Heddle 1991; Hendricks et al. 1985; Rice et al. 2000; Wu et al. 2003) and three of the studies reviewed evaluated the toxicity of a mixture of PAHs to juvenile chinook salmon (Meador 1997; Palm et al. 2003; Rice et al. 2000). Separate TRVs were selected for benzo(a)pyrene and the PAH mixture for comparison to benzo(a)pyrene and total PAHs exposure data, respectively in the Round 2 ERA.

Benzo(a)pyrene

Table 2 summarizes the feeding rate assumptions, dietary dose calculations, and dietary NOAELs and LOAELs for benzo(a)pyrene reported in the reviewed literature. Three of the four studies reported either a feeding rate or an exposure rate (Hendricks et al. 1985; Rice et al. 2000; Wu et al. 2003). Hart and Heddle (1991) did not present an exposure rate or feeding rate so a feeding rate of 1.9% body weight/day was assumed based on the average feeding rate of rainbow trout fed a synthetic diet in 16 dietary toxicity studies (Baldisserotto et al. 2005; Cockell and Bettger 1993; Cockell and Hilton 1988; Cockell et al. 1991, 1992; Franklin et al. 2005; Galvez and Wood 1999; Handy 1993a, b; Hendricks et al. 1985; Kamunde et al. 2001; Lanno et al. 1985a, b; Macek et al. 1970; Oladimeji et al. 1984; Rodgers and Beamish 1982). Rice et al.(2000) presented a length but did not present a body weight so the juvenile English sole weight was estimated assuming the length:weight relationship of English sole collected from the Lower Duwamish Waterway in Seattle, Washington (Luxon 2006).

Dietary dose LOAELs ranged from 1.4 mg/kg bw/day for growth of English sole (Rice et al. 2000) to 19 mg/kg bw/day for growth of rainbow trout (Hart and Heddle 1991). At the lowest LOAEL, Rice et al. (2000) exposed juvenile English sole to live polychaete worms that had been previously exposed to sediments spiked with benzo(a)pyrene at two different doses. Reduced daily growth rate was observed at the high dose of 1.4 mg/kg bw/day but not the low dose 0.66 mg/kg bw/day. The LOAEL TRV of 1.4 mg/kg bw/day was selected because this was the lowest LOAEL reported

in the reviewed literature. Dietary NOAELs ranged from 0.66 mg/kg bw/day for growth of English sole (Rice et al. 2000) to 12.5 mg/kg bw/day for growth and survival of areolated grouper (Wu et al. 2003). The NOAEL of 0.66 mg/kg bw/day reported in Rice et al. (2000) was selected as the NOAEL TRV. There is uncertainty associated with the selected TRVs because the literature dataset for the dietary toxicity of benzo(a)pyrene to fish is limited to four studies. Estimation of the body weight for English sole does not affect the dose calculation and does not affect the uncertainty of the TRVs.

PAH Mixture

Three studies were identified in which fish were exposed to a mixture of PAHs (Rice et al. 2000; Palm et al. 2003; Meador et al. 2006). Adverse effects on growth were evaluated for two species of fish (juvenile English sole and juvenile chinook salmon). Table 2 summarizes the feeding rate assumptions, dietary dose calculations, and dietary NOAELs and LOAELs for DDT reported in the reviewed literature. All three studies reported a dietary dose or a feeding rate from which a dietary dose could be calculated.

Dietary dose LOAELs ranged from 0.12 mg/kg bw/day for growth of English sole (Rice et al. 2000) to 18 mg/kg bw/day for growth of juvenile chinook salmon (Meador et al. 2006). At the lowest LOAEL, Rice et al. (2000) exposed juvenile English sole to polychaete worms that had been previously exposed in the laboratory to 0.1% field contaminated sediments from the Eagle Harbor, Washington, Superfund site mixed with 99% sediments from a reference site. Only PAH concentrations were reported for the sediment and worm tissue, however, other uncharacterized chemicals may also have been present. Fish exposed to contaminated worms at a total PAH dose of 0.12 mg/kg bw/day had a lower daily growth rate than controls. In a second trial of the experiment at the same dietary concentration and a slightly higher feeding rate, a similar trend was observed but the effect on growth was not statistically significant. This LOAEL was not selected because the worms used were exposed to field-collected sediments with uncharacterized chemicals.

At the next lower LOAEL, Meador et al. (2006) exposed juvenile chinook salmon to total PAHs dietary doses of 0.026 (in the control group), 0.7, 2.3, 6.1, 18, or 22.1 mg/kg bw/day for 53 days following smoltification. The mixture was of 21 PAHs based on the mixture of PAHs observed in juvenile chinook stomach contents from the Duwamish River waterway (Varanasi et al. 1993) Adverse effects on the wet weight of the fish were observed in the highest dose relative to the control and adverse effects on the dry weight of the fish were observed for the two highest concentrations relative to the control. Fish from the two highest doses also had significantly higher ww to dw ratios. Weights of PAH dosed fish at all concentrations were significantly more variable than weights of control fish, however, this effect wasn't dose-responsive. Based on these results, a LOAEL of 18.0 mg/kg bw/day for growth on a dw basis was selected as the LOAEL TRV because this was the only LOAEL reported in the two studies reviewed. The NOAEL of 6.1 mg/kg bw/day for growth on a dw basis from

this study was also selected. There is high uncertainty associated with the selected TRVs because the literature dataset for the dietary toxicity of PAH mixtures to fish is limited to three studies.

2.2.3 DDT, Mercury, and PCBs

Dietary TRVs were developed for DDT, mercury, and PCBs, as presented in the following subsections.

DDT

Eleven studies measuring the effects of dietary DDT to fish (Berlin et al. 1981; Buhler et al. 1969; Butler 1969; Hilton et al. 1983; Jarvinen et al. 1976, 1977; Macek 1968a, b; Macek and Korn 1970; Macek et al. 1970; Rhead and Perkins 1984; Warlen et al. 1977) were evaluated for TRV selection. Effects on growth, survival, behavior, and reproduction were reported in eight fish species (including Atlantic menhaden, brook trout, goldfish, pinfish, rainbow trout, chinook salmon, fathead minnow, and coho salmon) following exposure to dietary DDT.

Table 2 summarizes the feeding rate assumptions, dietary dose calculations, and dietary NOAELs and LOAELs for DDT reported in the reviewed literature. Seven of the eleven dietary TRV papers presented either feeding rate or exposure rate data (Buhler et al. 1969; Butler 1969; Macek 1968a, b; Macek and Korn 1970; Macek et al. 1970; Warlen et al. 1977). Four studies did not present a feeding rate or dose (Hilton et al. 1983; Rhead and Perkins 1984; Berlin et al. 1981; Jarvinen et al. 1976, 1977)[3] so a feeding rate was estimated from the available literature for each fish species. A feeding rate of 2% body weight per day was assumed for Rhead and Perkins (1984) as a typical feeding rate used in dietary toxicity studies for fish because no goldfish specific dietary toxicity studies were identified. An average feeding rate of 1.7% for brook trout fed a synthetic diet based on three dietary toxicity studies (Macek and Korn 1970; Macek 1968a, b) was assumed for Berlin et al. (1981). A feeding rate of 1.9 % bw/day was assumed for Hilton et al. (1983) based on the average feeding rate of rainbow trout fed a synthetic diet in 16 dietary toxicity studies (Baldisserotto et al. 2005; Cockell and Bettger 1993; Cockell and Hilton 1988; Cockell et al. 1991, 1992; Franklin et al. 2005; Galvez and Wood 1999; Handy 1993a, b; Hendricks et al. 1985; Kamunde et al. 2001; Lanno et al. 1985a, b; Macek et al. 1970; Oladimeji et al. 1984; Rodgers and Beamish 1982). A feeding rate of 6.0 % bw/day was assumed for dietary exposure to fathead minnow in the study by Jarvinen et al. (1976; 1977) based on a fathead minnow feeding rate for a pelletized food diet reported in Ogle and Knight (1989).

Dietary dose LOAELs ranged from 0.0044 mg/kg bw/day for mortality of brook trout (Berlin et al. 1981), to 3.0 mg/kg bw/day for mortality of juvenile chinook salmon (Buhler et al. 1969). At the lowest LOAEL of 0.0044 mg/kg bw/day, Berlin et al.

[3] Note that Jarvinen et al. 1976 and 1977 report the results of the same study.

(1981) exposed fish grown from field-collected eggs to various dietary concentrations of DDE for 176 days, and fry mortality was observed at all exposure levels. The field-collected eggs contained 7.6 mg/kg of PCB and 4.7 mg/kg of DDE in the tissue and possibly other, uncharacterized organic chemicals that could have contributed to the reported toxicity, and the eggs were exposed to aqueous DDE as well as dietary DDE. At the next lowest LOAEL of 0.14 mg/kg bw/day, embryo survival was reduced in brook trout following a 156-day exposure of adults (prior to spawning) to pelleted fish food treated with DDT (Macek 1968b). Dietary dose NOAELs ranged from 0.0023 mg/kg bw/day for growth of Atlantic menhaden (Warlen et al. 1977) to 0.75 mg/kg bw/day for mortality of chinook salmon (Buhler et al. 1969).The LOAEL of 0.14 mg/kg bw/day (Macek 1968b) was selected as the dietary dose LOAEL TRV. This selected LOAEL represents the lowest LOAEL reported in the literature reviewed in which fish were exposed to DDT only. No NOAEL based on reproduction was reported in the reviewed studies, so none was selected.

Mercury

Five studies measuring the effects of dietary mercury on fish were evaluated for TRV selection (Friedmann et al. 1996; Hammerschmidt et al. 2002; Matta et al. 2001; Rodgers and Beamish 1982; Webber and Haines 2003)). Effects on growth, survival, reproduction, and behavior were reported in five fish species (fathead minnow, golden shiner, walleye, mummichog, and rainbow trout) following exposure to dietary mercury. Table 2 summarizes the feeding rate assumptions, dietary dose calculations, and dietary NOAELs and LOAELs for mercury reported in the reviewed literature. Three of the five dietary toxicity studies presented either exposure rate or feeding rate data from which exposure rates could be directly calculated (Friedmann et al. 1996; Rodgers and Beamish 1982; Webber and Haines 2003). Two studies did not present sufficient information to calculate a feeding rate (Matta et al. 2001; Hammerschmidt et al. 2002). A feeding rate of 2.5% bw/day was assumed for dietary exposure to mummichog in the study by Matta et al. (2001) based on a mummichog feeding rate for a pelletized food diet reported in Gutjahr-Gobell et al. (1999). A feeding rate of 6.0% bw/day was assumed for dietary exposure to fathead minnow in the study by Hammerschmidt et al. (2002) based on a fathead minnow feeding rate for a pelletized food diet reported in Ogle and Knight (1989).

Dietary dose LOAELs ranged from 0.0091 mg/kg bw/day for behavior of golden shiner (Webber and Haines 2003) to 1.5 mg/kg bw/day for behavior of rainbow trout (Rodgers and Beamish 1982). At the lowest LOAEL of 0.0091 mg/kg bw/day, Webber and Haines (2003) reported increased shoal area following exposure to a model predator compared to controls. At this dose, no effects were observed for two other predation avoidance behaviors, shoal height, and time to settling. The authors state that these behaviors may increase the chance of predation but note that this has not been confirmed in wild fish. Because these behavioral effects cannot be tied directly to effects on growth, mortality, or reproduction, this study was not selected to derive TRVs for the Round 2 ERA. At the next lowest LOAEL of 0.048 mg/kg bw/day, Matta et al. (2001) exposed mummichog to various concentrations of dietary methyl

mercury for at least 6 weeks prior to spawning and effects on survival, growth, and reproduction were evaluated. Increased mortality was observed in male mummichog at a dietary dose of 0.048 mg/kg bw/day. This LOAEL was selected because it is the lowest LOAEL where adverse effects on growth, mortality, or reproduction were observed.

Dietary dose NOAELs ranged from 0.0036 mg/kg bw/day for growth and gonadal development of juvenile walleye (Friedmann et al. 1996) to 1.5 mg/kg bw/day for mortality of rainbow trout (Rodgers and Beamish 1982). The NOAEL of 0.013 mg/kg bw/day reported in Matta et al. (2001), where no effect on male mummichog survival was observed was selected as the NOAEL TRV. The selected NOAEL and LOAEL TRVs are highly uncertain because the dietary doses were based on a FIR for mummichog reported in a dietary toxicity study (Gutjahr-Gobell et al. 1999) that was unrelated to the study that the TRVs were based on (Matta et al. 2001).

PCBs

Nine toxicity studies were evaluated for TRV selection that measured the effects of dietary PCBs to fish (Berlin et al. 1981; Hansen et al. 1976; Hendricks et al. 1981; Hugla and Thome 1999; Lieb et al. 1974; Matta et al. 2001; Mayer et al. 1977; McCarthy et al. 2003; Powell et al. 2003). The effects on growth, survival, and reproduction were measured following exposure to dietary PCBs in eight fish species (brook trout, barbels, rainbow trout, Atlantic croaker, chinook salmon, coho salmon, channel catfish, and mummichog).

Table 2 summarizes the feeding rate assumptions, dietary dose calculations, and dietary NOAELs and LOAELs for PCBs derived from the reviewed literature. Five of the nine dietary toxicity studies presented either exposure rate or feeding rate data from which exposure rates could be directly calculated (Hansen et al. 1976; Lieb et al. 1974; Mayer et al. 1977; McCarthy et al. 2003; Powell et al. 2003). Four studies did not present sufficient information to estimate a feeding rate (Berlin et al. 1981; Matta et al. 2001; Hendricks et al. 1981; Hugla and Thome 1999). A feeding rate of 1.7% bw/day was assumed for Berlin et al. (1981) based on the average feeding rate of brook trout in 3 dietary toxicity studies (Macek 1968b, a; Macek and Korn 1970). A feeding rate of 1.9% was estimated for Hendricks et al. (1981) based on the average feeding rate of rainbow trout fed a synthetic diet in 16 dietary toxicity studies (Baldisserotto et al. 2005; Cockell and Bettger 1993; Cockell and Hilton 1988; Cockell et al. 1991, 1992; Franklin et al. 2005; Galvez and Wood 1999; Handy 1993a, b; Hendricks et al. 1985; Kamunde et al. 2001; Lanno et al. 1985a, b; Macek et al. 1970; Oladimeji et al. 1984; Rodgers and Beamish 1982). A feeding rate of 2.5% bw/day was assumed for Matta et al. (2001) based on a mummichog feeding rate for a pelletized food diet reported in Gutjahr-Gobell et al. (1999). A feeding rate of 2% bw/day was assumed for Hugla and Thome (1999) because no feeding rate data for barbel could be identified. A body weight of 1 kg was assumed.

Dietary dose LOAELs ranged from 0.018 mg/kg bw/day for mortality of brook trout (Berlin et al. 1981), to 14.5 mg/kg bw/day for mortality of coho salmon (Mayer et al. 1977). At the lowest LOAEL, Berlin et al. (1981) exposed field-collected eggs to three levels of PCB concentrations via diet and water for 176 days, and fry mortality was observed at all exposure levels. However, the field-collected eggs contained 7.6 mg/kg of PCB and 4.7 mg/kg of DDE in the tissue and possibly other uncharacterized organic chemicals that could have contributed to the reported toxicity, and eggs were exposed to both aqueous PCBs and dietary PCBs. Therefore, the next lowest LOAEL was evaluated for TRV selection. At the next lower LOAEL, Hugla and Thome (1999) exposed adult barbell to two doses of a dietary mixture of Aroclor 1254 and 1260 and evaluated effects on fertility and reproductive success. Fish administered the low dose of 0.050 mg/kg bw/day had reduced fecundity compared with controls. NOAELs ranged from 0.34 mg/kg bw/day for growth of juvenile chinook salmon (Powell et al. 2003) to 1.5 mg/kg bw/day for mortality of coho salmon (Mayer et al. 1977).

The LOAEL of 0.050 mg/kg bw/day (Hugla and Thome 1999) was selected as the dietary dose LOAEL TRV for PCBs in fish. This LOAEL represents the lowest LOAEL reported in the literature reviewed in which fish were exposed to PCBs only. The selected LOAEL TRV is highly uncertain because the feeding rate used to derive the dietary dose LOAEL is assumed based on typical feeding rates used for fish dietary toxicity studies. No NOAEL based on reproduction that was lower than the selected LOAEL was reported in the reviewed studies and none was selected.

3.0 ROUND 2 WILDLIFE TRVS

Selected bird and mammal dietary TRVs were updated from Appendix B of the Ecological PRE (Windward 2005) for some chemicals, based on additional toxicological studies identified and reviewed. These TRVs were used in the Round 2 ERA. Updated selected bird and mammal dietary TRVs from the Round 2 ERA are summarized in the following subsections.

3.1 BIRD DIETARY TRVS

Additional avian toxicological literature was reviewed and/or changes to bird TRV calculations were made for nine chemicals: PCBs, cadmium, chromium, copper, lead, selenium, zinc, total PAHs, aldrin, and methoxychlor. Table 3 summarizes the feeding rate assumptions, dietary dose calculations, and dietary NOAELs and LOAELs for these dietary dose bird TRVs.

3.1.1 PCBs

Three additional studies were reviewed and evaluated for TRV selection on the oral toxicity of PCB Aroclors to birds (Fernie et al. 2003a, b, c). However, the selected LOAEL and NOAEL TRV did not change from Appendix B of the Ecological PRE (Windward 2005).

3.1.2 Metals

Cadmium

Three additional studies were reviewed and evaluated for TRV selection on the oral toxicity of cadmium to birds (Freeland and Cousins 1973; Pritzl et al. 1974; Richardson et al. 1974). In addition, the LOAEL and NOAELs from Leach et al. (1979) were recalculated using a food ingestion rate of 0.1019 kg/day based on the National Research Council (1984). The updated NOAEL and LOAEL TRV derived from Leach et al. (1979) of 2.9 and 0.73 mg/kg bw/day, respectively, were selected as the TRVs for cadmium. The selected LOAEL of 2.9 mg/kg bw/day was based on the study resulting in the lowest LOAEL where a clear dose-response relationship was observed (Leach et al. 1979).

There is high uncertainty associated with the selected TRVs for cadmium. The selected TRVs are based on eggshell thinning in chickens. Egg production in a domestic species is not an appropriate endpoint for TRV selection; and it is unknown if the degree to which eggshell thinning observed in chickens would affect other non-domestic species and reproductive success.

Chromium

One additional study that exposed birds to dietary chromium (Lien et al. 2004) was evaluated for TRV selection. The lowest NOAEL of 0.10 mg/kg bw/day was derived from Lien et al (2004) where no effect on eggshell weight and shell thickness was reported in domestic chickens; however, the effect of eggshell thinning on reproductive success (e.g., hatchability or offspring viability) was not reported, therefore, the selected LOAEL and NOAEL TRV did not change from Appendix B of the Ecological PRE.

Copper

Four additional studies on the toxicity of dietary copper to birds (Balevi and Coskun 2004; Dozier et al. 2003; Lien et al. 2004; Persia et al. 2004) was evaluated for TRV selection. However, the selected LOAEL and NOAEL TRV for copper did not change from Appendix B of the Ecological PRE.

Lead

One additional study was reviewed on the potential adverse effects of lead on birds (Morgan et al. 1975). However, the selected LOAEL and NOAEL TRV for lead did not change from Appendix B of the Ecological PRE.

Selenium

One additional study that exposed birds to selenium through diet were reviewed and evaluated for TRV selection (Choct et al. 2004). However, the selected LOAEL and NOAEL TRV for selenium did not change from Appendix B of the Ecological PRE.

Zinc

Two additional studies on zinc toxicity to birds (Dozier et al. 2003; Persia et al. 2004) were evaluated for TRV selection. However, the selected LOAEL and NOAEL TRV for selenium did not change from Appendix B of the Ecological PRE.

3.1.3 Pesticides

Aldrin

Aldrin was identified as a wildlife chemical of interest (COI) in the Round 2 ERA. Two studies that measured the toxicity of aldrin to birds (DeWitt 1956; Hall et al. 1971) were reviewed and evaluated for TRV selection. Adverse effects on growth or survival of domestic species (quail and pheasant) were observed following oral exposure to aldrin. LOAELs ranged from 0.04 mg/kg bw/day for mortality of quail (DeWitt 1956) to 92 mg/kg bw/day for mortality of pheasant chicks (Hall et al. 1971). At the lowest LOAEL, 97.5% mortality was observed after 127 days in quails following dietary exposure of aldrin. Only one NOAEL of 10 mg/kg bw/day was derived from the reviewed studies. At this NOAEL, no effect on pheasant chicks growth was observed following seven weekly exposure doses of a gelatin capsule treated with aldrin.

The lowest LOAEL calculated from the reviewed literature of 0.04 mg/kg bw/day was selected as the TRV for aldrin. There were no NOAELs lower than the selected LOAEL reported in the literature reviewed, so none was selected. There is some uncertainty associated with the selected LOAEL TRV because the literature dataset for aldrin toxicity in birds is limited to two studies reporting the effects in domestic species only, with high variability in the dose levels resulting in adverse effects. The sensitivity of other bird species to aldrin is unknown.

Methoxychlor

Two additional studies that measured the toxicity of methoxychlor to birds (Gee et al. 2004; Millam et al. 2002) were reviewed and evaluated for TRV selection. In these two studies, the effects of oral methoxychlor on survival and reproduction in zebra finches is reported. Only one LOAEL was reported in the studies reviewed; at 346 mg/kg bw/day, significant mortality and affect on hatchability of zebra finches (Gee et al. 2004; Millam et al. 2002) was observed following exposure to methoxychlor via oral gavage for one week. No effect was reported in zebra finches fed 34.6 mg/kg bw/day. The reported LOAEL and NOAEL of 346 and 34.6 mg/kg bw/day from these twp studies, were selected as the TRVs for methoxychlor. There is uncertainty associated with the selected TRVs because they are based on oral gavage exposure. There is additional uncertainty because the literature dataset for methoxychlor toxicity in birds is limited.

3.1.3 Total PAHs

A benzo(a)pyrene TRV was developed and selected in Appendix B of the Ecological PRE. For the Round 2 ERA, a PAH mixture TRV was also developed. One study was reviewed and evaluated for TRV selection on the toxicity of a PAH mixture to birds (Patton and Dieter 1980). At 40 mg/kg bw/day, growth was adversely affected in mallards fed an petroleum hydrocarbon mixture containing PAHs[4] combined with paraffin wax at three months, however, at seven months, growth was recovered and change in body weight was not significantly different than the control group (Patton and Dieter 1980). The reduction observed in growth at three months was attributed to food avoidance of the aromatic hydrocarbon mixture feed because of the noxious odor of the petroleum hydrocarbons. This dose (40 mg/kg bw/day) was selected as the NOAEL TRV for total PAHs for birds. No LOAEL TRV was reported in this study, and none was selected.

There is very high uncertainty associated with the selected total PAH NOAEL TRV for birds based on the limited toxicological data (one study) reviewed. The selected NOAEL TRV is based on exposure to a petroleum hydrocarbon mixture that contained some individual PAHs. Therefore, birds were exposed to a PAH mixture, in addition

[4] Aromatic hydrocarbon mixture contained the following: ethylbenzene, 1,2,3,4-tetrahydronaphthalene, dimethylnaphthalene, 2,3,3-trimethylindolenine, acenaphthylene, acenaphthlene, phenanthrene, 2-methylbenzothiazole, dibenzothiophene, and 2,6-dimethylquinoline.

to other chemicals. In addition, food avoidance was observed in the mixture PAH study and no LOAEL TRV was derived.

3.2 ROUND 2 MAMMAL DIETARY TRVS

Additional mammalian toxicological literature was reviewed and/or changes to mammal TRV calculations were made for six chemicals: selenium, zinc, dibutyltin, tributyltin, aldrin, and dieldrin. Table 3 summarizes the feeding rate assumptions, dietary dose calculations, and dietary NOAELs and LOAELs for these dietary dose mammal TRVs.

3.2.1 Metals

Selenium

Two additional laboratory studies on the toxicity of selenium to mammals were evaluated for TRV selection (Behne et al. 1992; Jia et al. 2005). In addition, the LOAEL and NOAELs from Halverson et al. (Halverson et al. 1966) were updated based on to reflect the lowest concentration in the diet were adverse effects were observed (effects were not statistically evaluated). The updated NOAEL and LOAEL TRV derived from Halverson et al. (1966) of 0.08 and 0.055 mg/kg bw/day, respectively, were selected as the TRVs for selenium. These were the lowest TRVs calculated in the literature reviewed.

Zinc

No additional studies on dietary toxicity of zinc to mammals were identified. However, the selected NOAEL and LOAEL from Schlicker and Cox (1968) were recalculated using a body weight of 0.35 kg, consistent with Sample (1996). The selected LOAEL was based on the recalculated LOAEL TRV of 320 mg/kg bw/day derived from Schlicker and Cox (1968) where significantly reduced fetal growth and an increased number of fetal resorptions in rats fed zinc as zinc oxide during gestation was observed. The selected LOAEL represents the lowest LOAEL calculated from a study where effects were statistically evaluated. The recalculated NOAEL derived from this study of 160 mg/kg bw/day was selected as the NOAEL TRV for zinc.

Dibutyltin

Dibutyltin was identified as a wildlife COI in the Round 2 ERA. Two studies on the toxicity of dibutyltin to mammals were evaluated for TRV selection (Ema et al. 2003; Harazono and Ema 2003). In both studies, rats were exposed to dibutyltin via gavage for three days during gestation. A LOAEL of 7.6 mg/kg bw/day was reported in both studies. At this LOAEL, maternal body weight, and reproductive success (as measured by number of implantations, number of pregnant females, and rate of post-implantation loss) was reduced. A NOAEL of 3.8 mg/kg bw/day was reported in Harazono and Ema (2003), where no effect on maternal body weight was observed.

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The only LOAEL and NOAEL, 7.6 and 3.8 mg/kg bw/day, respectively, were selected was the dibutyltin TRVs for mammals. There is uncertainty associated with the selected LOAEL because gavage is not a preferred exposure route; however, no dietary studies were identified in the literature. There is additional uncertainty associated with the selected TRVs because the literature dataset for dibutyltin toxicity in mammals is limited to two studies.

Tributyltin

Four additional that exposed mammals to dietary TBT were reviewed and evaluated for TRV selection (Omura et al. 2001; Makita et al. 2003; Ogata et al. 2001).At the lowest LOAEL of 2.0, pup body weight was reduced in the offspring of adult rats fed 2.0 mg/kg bw/day TBT as tributyltin chloride during a reproductive period (Omura et al. 2001). The lowest LOAEL of 2.0 mg/kg bw/day was selected as the LOAEL TRV for TBT in mammals. This was the lowest TRV calculated from or reported in the reviewed literature and was based a dietary study. The NOAEL reported in the same study, 0.4 mg/kg bw/day was selected as the NOAEL TRV.

3.2.2 Pesticides

Aldrin

Aldrin was identified as a wildlife COI in the Round 2 ERA. One study on the toxicity of aldrin to mammals was reviewed and evaluated for TRV selection (Reuber 1980). Increased mortality was reported in rats exposed to 4.1 mg/kg bw/day dietary aldrin over two years. No effect on survival was observed in rats following exposure to 0.83 mg/kg bw/day. The LOAEL and NOAEL of 4.1 and 0.83 mg/kg bw/day, respectively, were selected as the TRVs. There is some uncertainty associated with the selected TRVs because the literature dataset for aldrin toxicity to mammals is limited to one study.

Dieldrin

One additional study that measured the dietary toxicity of dieldrin to mammals were evaluated for TRV selection (Reuber 1980). However, the selected LOAEL and NOAEL TRV for dieldrin did not change from Appendix B of the Ecological PRE.

Methoxychlor

One additional study on the effects of methoxychlor on mammals were reviewed and evaluated for TRV selection (Reuber 1980). However, the selected LOAEL and NOAEL TRV for methoxychlor did not change from Appendix B of the Ecological PRE.

3.2.3 SVOCs

Phenol

Three studies, as cited in the US Environmental Protection Agency's (EPA's) online database, Integrated Risk Information System (IRIS) (EPA 2006) were summarized on

the toxicity of phenol toxicity to rats or mice were evaluated for TRV selection (Argus Research Laboratories 1997; Charles River Laboratories 1988; NTP 1983a, b; all as cited in IRIS [EPA 2006]). Observed adverse effects, including various reproductive parameters and growth, were reported in laboratory rats and mice following exposure to phenol via oral gavage. LOAELs ranged from 120 mg/kg bw/day for reproduction and growth of rats (Argus Research Laboratories 1997 and Charles River Laboratories 1988 to 280 mg/kg bw/day for survival and growth of mice (NTP 1983b). At the lowest LOAEL of 120 mg/kg bw/day, reduced maternal body weight and fetal body weight were observed following exposure to phenol administered via oral gavage during gestation. NOAELs ranged from 60 mg/kg bw/day for reproduction and growth of rats (Argus Research Laboratories 1997; Charles River Laboratories 1988) to 140 mg/kg bw/day for survival and growth of mice (NTP 1983b).

The lowest LOAEL and NOAEL of 120 and 60 mg/kg bw/day, respectively, were selected as the dietary mammal TRVs for phenol. There is uncertainty associated with the selected TRVs because they are based on oral gavage exposure and because the literature dataset for phenol toxicity in birds is limited to three studies. There is additional uncertainty associated with the selected TRVs because the original studies couldn't be obtained for review.

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TABLES

Table 1. Updated Round 2 ERA Fish and Wildlife TRVs

Chemical	NOAEL	LOAEL	Test Species	Endpoint	Exposure Duration	Source	Reason for Change in TRV
Fish Tissue TRVs (mg/kg ww)							
PCBs	1.9	--	sheepshead minnow (adult)	decreased fry survival	maternal exposure in water for 28 days	Hansen et al. (1973)	change in TRV selection criteria for Round 2 ERA
	--	4.02[a]	Atlantic salmon	reduced live fry body weight	water for 48 hours	Fisher et al. (1994)	
Mercury	0.2	0.47	mummichog	reduced male survival	water for 42 days	Matta et al. (2001)	change in TRV selection criteria for Round 2 ERA
Fish Dietary Dose TRVs (mg/kg bw/day)							
Arsenic	0.40	0.60	rainbow trout (juvenile)	reduced body weight	fed treated food 8 weeks	Oladameji et al. (1984)	change in dietary assumptions
Cadmium	NS	0.010	rockfish (juvenile)	reduced body weight and length, growth rate, and condition factor	fed treated food 60 days	Kim et al. (2004) Kang et al. (2005)	change in dietary assumptions
Copper	0.24	0.48	channel catfish (fingerling)	reduced growth	fed treated food 16 weeks	Murai et al. (1981)	change in dietary assumptions
Lead	134	NA	rainbow trout (immature)	reduced growth	fed treated food 191 days	Goettl et al. (1976)	change in dietary assumptions
Mercury	0.013	0.048	mummichog	reduced male survival	fed treated food at least 6 weeks	Matta et al. (2001)	change in dietary assumptions
Selenium	0.10	0.20	bluegill (juvenile)	reduced survival	fed treated food 90 days	Cleveland et al. (1993)	change in dietary assumptions
Silver	70	NA	rainbow trout	no effect on growth	fed treated food 58 days	Galvez and Wood (1999)	change in dietary assumptions
Zinc	19.0	38.0	rainbow trout (fingerling)	reduced growth	fed treated food 6 weeks	Takeda and Shimma (1977)	change in dietary assumptions
TBT	0.0021	0.020	Japanese flounder (larvae)	reduced body weight, increased sex reversal	fed treated food 65 days	Shimasaki et al. (2003)	change in dietary assumptions

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Table 1. Updated Round 2 ERA Fish and Wildlife TRVs

Chemical	NOAEL	LOAEL	Test Species	Endpoint	Exposure Duration	Source	Reason for Change in TRV
Benzo(a) pyrene	0.66	1.4	English sole (juvenile)	reduced daily growth rate	fed contaminated worms for 10-12 days	Rice et al. (2000)	change in dietary assumptions
Total PAHs	6.1	18	rainbow trout (juvenile)	reduced body weight (as dry weight)	fed treated food 53 days	Meador et al. (2006)	change in dietary assumptions
Total DDT	NS	0.14	brook trout (yearling)	reduced embryo survival	fed treated food 156 days	Macek (1968b)	change in dietary assumptions
Total PCBs	NS	0.050	barbel	reduced fecundity, ethyoxyresorufin-o-deethylase induction	fed treated food 50 days	Hugla and Thome (1999)	change in dietary assumptions
Bird Dietary Dose TRVs (mg/kg bw/day)							
Cadmium	0.73	2.9	chicken	reduced egg production and eggshell thickness	48 weeks	Leach et al. (1979)	change in dietary assumptions
Total PAHs	40	NA	mallard	no effect on growth after duration of study	7 months	Patton and Dieter (1980)	addition of total PAH TRV for Round 2 ERA
Aldrin	NS	0.04	quail	reduced survival and no. eggs hatched, increased no. broken eggs	5 months	DeWitt (1956)	new COI for Round 2 ERA
Methoxychlor	34.6	346	zebra finch	2.5% survival	1 week	Gee et al. (2004); Millam et al. (2002)	additional literature reviewed resulted in lower TRVs
Mammal Dietary Dose TRVs (mg/kg bw/day)							
Selenium	0.055	0.08	rat	reduced body weight	fed treated food for 6 weeks	Halverson et al. (1966)	change to lowest dose where effect observed
Zinc	160	320	rat	reduced fetal growth and increased no. resorptions	fed treated food during gestation	Schlicker and Cox (1968)	change in dietary assumptions
Dibutyltin	3.8	7.6	rat	reduced growth and reproductive success	oral gavage during gestation	Harazono and Ema (2003); Ema et al. (2003)	new COI for Round 2 ERA
Tributyltin	0.4	2.0	rat	reduced offspring pup body weight	fed treated food for multiple generations	Omura et al. (2001); Makita et al. (2003); Makita et al. (2004)	additional literature reviewed resulted in lower TRVs

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Table 1. Updated Round 2 ERA Fish and Wildlife TRVs

Chemical	NOAEL	LOAEL	Test Species	Endpoint	Exposure Duration	Source	Reason for Change in TRV
Aldrin	0.83	4.1	rat	reduced survival	fed treated food for 2 years	Reuber (1980)	new COI for Round 2 ERA
Phenol	60	120	rat	reduced maternal and fetal body weight	oral gavage during gestation	Argus Research Laboratories 1997 and Charles River Laboratories 1988 (as cited in IRIS (EPA 2006))	toxicological literature found in additional literature search

[a] Estimated from an embryo concentration using an egg to adult conversion factors

bw – body weight

COI – chemical of interest

ERA – ecological risk assessment

PAH – polycyclic aromatic hydrocarbon

LOAEL – lowest-observed-adverse-effects-level

NA – No LOAEL TRV selected; no appropriate literature available

NOAEL – no-observed-adverse-effects-level

NS – NO NOAEL TRV selected; no NOAEL was available from the reviewed literature that was lower than the selected LOAEL TRV. Uncertainty factors were used in the Round 2 ERA to extrapolate NOAELs from unbounded selected LOAELs.

PCB – polychlorinated biphenyl

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Table 2. Summary of Updated Fish Dietary TRV Review for Round 2 ERA

Analyte	Chemical Form	NOAEL (mg/kg BW/day)	LOAEL (mg/kg BW/day)	Source	Body weight (kg)	FIR (kg-food/day)	NOAEL exposure concentration (mg/kg dw)	LOAEL exposure concentration (mg/kg dw)	Test Species	Lifestage	Endpoint	Endpoint Effect	Exposure Duration	Notes
Arsenic	disodium arsenate heptahydrate	0.12		Cockell et al. 1991	0.018	0.00027	8		Rainbow trout		growth	body weight	16 days	feed refusal accompanied effects; dose calculated from reported dietary concentration and feeding rate; reported dose = 0.153 mg/kg bw/day
Arsenic	disodium arsenate heptahydrate	0.21		Cockell et al. 1991	0.014	0.00025	12		Rainbow trout		growth	body weight gain	24 days	body weight gain reduced at 12 weeks, but not at 24 weeks; dose calculated from reported dietary concentration and feeding rate; reported dose = 0.281 mg/kg bw/day
Arsenic	**sodium arsenite**	**0.40**		**Oladimeji et al. 1984**	**0.039**	**0.00077**	**20**		**Rainbow trout**	**juvenile**	**growth**	**body weight**	**8 weeks**	**concentrations in figure and text do not agree in study: 20 mg/kg is mentioned both as an effect level and a NOAEL in the text; however, it is shown in the figure to be not significant. 20 mg/kg = assumed NOAEL**
Arsenic	disodium arsenate heptahydrate		0.50	Cockell et al. 1991	0.083	0.00084		49	Rainbow trout		growth	body weight	24 days	Mortality was greater in control; NOAEL supported by additional 9-week study; casein diet; dose calculated from reported dietary concentration and feeding rate; reported dose = 1.9 mg/kg bw/day.
Arsenic	disodium arsenate heptahydrate	0.52		Blazer et al. 1997	0.0521	0.00052	52.3		Striped bass	juvenile	growth	body weight	6 weeks	feed refusal accompanied effects
Arsenic	disodium arsenate heptahydrate	0.55		Cockell et al. 1992	0.0423	0.00073	32		Rainbow trout		growth	body weight	12 weeks	Dose calculated from reported dietary concentration and feeding rate; reported dose = 0.329 mg/kg bw/day
Arsenic	**sodium arsenite**		**0.60**	**Oladimeji et al. 1984**	**0.037**	**0.00075**		**30**	**Rainbow trout**	**juvenile**	**growth**	**body weight**	**8 weeks**	**concentrations in figure and text do not agree in study: 20 mg/kg is mentioned both as an effect level and a NOAEL in the text; however, it is shown in the figure to be not significant. 20 mg/kg = assumed NOAEL**
Arsenic	disodium arsenate heptahydrate		0.62	Cockell et al. 1991	0.013	0.00018		44	Rainbow trout		growth	body weight	16 days	feed refusal accompanied effects; dose calculated from reported dietary concentration and feeding rate; reported dose = 0.583 mg/kg bw/day
Arsenic	disodium arsenate heptahydrate		0.63	Cockell et al. 1991	0.013	0.00025		33	Rainbow trout		growth	body weight gain	24 days	feed consumption did not differ from controls; body weight gain reduced at 12 weeks, but not at 24 weeks; dose calculated from reported dietary concentration and feeding rate; reported dose = 0.855 mg/kg bw/day
Arsenic	disodium arsenate		1.0	Cockell and Bettger 1993	0.047	0.00083		58	Rainbow trout		growth	body weight	4 weeks	feed refusal accompanied effects
Arsenic	disodium arsenate heptahydrate		1.0	Cockell et al. 1992	0.0299	0.00051		60	Rainbow trout		growth	body weight	12 weeks	feed refusal accompanied effects; dose calculated from reported dietary concentration and feeding rate; reported dose = 0.552 mg/kg bw/day
Arsenic	disodium arsenate heptahydrate		1.4	Cockell et al. 1992	0.0048	0.00012		55	Rainbow trout		growth	body weight	8 weeks	feed refusal accompanied effects; growth effects at LOAEL; dose calculated from reported dietary concentration and feeding rate; reported dose = 1.552 mg/kg bw/day
Arsenic	arsenic trioxide		1.7	Cockell and Hilton 1988	0.00459	0.000056		137	Rainbow trout		growth	body weight	8 weeks	feed refusal accompanied effects; greater than 10% mortality at LOAEL; study also reported effects of organic arsenic which was not toxic at high levels; dose calculated from reported dietary concentration and feeding rate; reported dose = 1.56 mg/kg bw/day
Arsenic	disodium arsenate heptahydrate		1.9	Blazer et al. 1997	0.0426	0.00043		188.8	Striped bass	juvenile	growth	body weight	6 weeks	feed refusal accompanied effects
Arsenic	disodium arsenate		2.9	Cockell and Hilton 1988	0.006575	0.00011		180	Rainbow trout		growth	body weight	8 weeks	Greater than 10% mortality at LOAEL; feed refusal accompanied effects; study also reported effects of organic arsenic which was not toxic at high levels; dose calculated from reported dietary concentration and feeding rate; reported dose = 2.97 mg/kg bw/day
Cadmium	**Cd(NO3)2**		**0.010**	**Kim et al. 2004; Kang et al. 2005**		**0.000494**	**0.05**	**0.5**	**rockfish**	**juvenile**		**body weight and length growth rate; condition factor**	**60 days**	
Cadmium		0.011		Maage 1990	0.000315	0.0000063	0.570		Atlantic salmon	fry	growth	body weight	4 wks	no control was used in study; fish were exposed to diet with copper, cadmium, and zinc; an increase in growth was observed at NOAEL; A feeding rate of 2% bw/ day was assumed because FIR was not reported; Not recommended for TRV selection

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Table 2. Summary of Updated Fish Dietary TRV Review for Round 2 ERA

Analyte	Chemical Form	NOAEL (mg/kg BW/day)	LOAEL (mg/kg BW/day)	Source	Body weight (kg)	FIR (kg-food/day)	NOAEL exposure concentration (mg/kg dw)	LOAEL exposure concentration (mg/kg dw)	Test Species	Lifestage	Endpoint	Endpoint Effect	Exposure Duration	Notes
Cadmium	cadmium chloride	1.0		Hatakeyama and Yasuno 1982	0.000859	0.0000372	171		guppy		growth		10-30 days	fed exposed Monia macrocopa, significant growth effect at day 10 disappeared at day 20 in fish fed 126 and 171 µg/g dw. Body weight from Hatekayama and Yasuno 1982.
Cadmium	cadmium chloride	1.2		Hatakeyama and Yasuno 1987	0.000859	0.0000372	210		guppy	adult	reproduc-tion	# live fry, fry mortality, premature embryos	2 months	fed exposed Chironomus yoshimatsui (midge larvae), no reproductive effect after 2 months.
Cadmium	cadmium chloride	1.6		Hatakeyama and Yasuno 1987	0.000859	0.0000372	274		guppy	2 months	growth	body weight	30 days	fed exposed Chironomus yoshimatsui (midge larvae), no growth effect by day 30.
Cadmium	Cd(NO3)2	2.5		Kim et al. 2004		0.000494	125		rockfish	juvenile	mortality		60 days	
Cadmium	cadmium chloride	3.2		Hatakeyama and Yasuno 1987	0.000779	0.0000372	500		guppy	30 day old	reproduc-tion	cummulative number of fry produced	7 months	fed exposed Chironomus yoshimatsui (midge larvae); cumulative number of fry decreased to about 60% of the control in fish exposed to 80 and 160 µg/L; no discussion of statistics; unclear on the reproductive effects at lower levels; LOAEL is estimated using figure, no replication.
Cadmium	cadmium chloride	4.1		Mount et al. 1994	0.000611	0.000037	69		Rainbow trout	fry	mortality	100% survival	60 days	Fish were exposed to copper, cadmium, lead, and zinc in water at 23.0, 0.97, 3.32, and 46.3 µg/L, respectively, at the same time as the dietary exposure; fish exposed to live prey
Cadmium	cadmium chloride		4.6	Hatakeyama and Yasuno 1987	0.000859	0.0000372		800	guppy	30 day old	reproduc-tion	cummulative number of fry produced	7 months	fed exposed Chironomus yoshimatsui (midge larvae); cumulative number of fry decreased to about 60% of the control in fish exposed to 80 and 160 µg/L; no discussion of statistics; unclear on the reproductive effects at lower levels; LOAEL is estimated using figure, no replication.
Cadmium	$Cd(NO_3)_2 \cdot 4H_2O$	5.9		Baldisserotto et al. 2005	0.017040308	0.000341	294		rainbow trout	juvenile		specific growth rate, survival	15-30 days	
Cadmium	cadmium chloride		9.4	Hatakeyama and Yasuno 1987	0.000659	0.0000364		1,250	guppy	30 day old	growth; mortality	female growth and mortality	7 months	fed exposed Chironomus yoshimatsui (midge larvae); female body weight decreased -68% of control at the 48th day in fish exposed to 160 µg/L; no discussion of statistics-unclear on growth effects at the other doses; LOAEL is estimated using figure; no effect on male growth, no replication.
Cadmium		9.4		Franklin et al. 2005	0.019	0.000380	471		rainbow trout	juvenile		growth rate, survival	28 days	dose calculated from reported dietary concentration and feeding rate; reported dose = 8.3 mg/kg bw/ day
Cadmium	$CdSO_4$		68	Handy 1993b	0.131	0.00089		10,000	Rainbow trout		mortality		28 days	39% (14 of 36) mortality between day 3 and 23 of exposure period
Copper	**copper**	**0.24**		**Murai et al. 1981**	**0.046**	**0.00137**	**8**		**Channel catfish**	**fingerling**	**growth**	**reduced growth**	**16 wks**	**significant effects at 4 weeks in 16- and 32-mg/kg treatments; growth gain per feed consumed was significantly lower for fish fed 8 mg/kg however considered NOAEL because BW not significantly different; FIR reported in Murai and Andrews 1978.**
Copper	**copper**		**0.48**	**Murai et al. 1981**	**0.036**	**0.00108**		**16**	**Channel catfish**	**fingerling**	**growth**	**reduced growth**	**16 wks**	**significant effects at 4 weeks in 16- and 32-mg/kg treatments; FIR reported in Murai and Andrews 1978.**
Copper	copper sulfate	1.0	2.0	Kang et al. 2005	0.031	0.00052	50	100	rockfish	juvenile		body weight (growth rate)	60 days	

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Table 2. Summary of Updated Fish Dietary TRV Review for Round 2 ERA

Analyte	Chemical Form	NOAEL (mg/kg BW/day)	LOAEL (mg/kg BW/day)	Source	Body weight (kg)	FIR (kg-food/day)	NOAEL exposure concentration (mg/kg dw)	LOAEL exposure concentration (mg/kg dw)	Test Species	Lifestage	Endpoint	Endpoint Effect	Exposure Duration	Notes
Copper	$CuSO_4$	2.2		Handy 1992 - Arch Env. Contam Toxicol. 22 parts 1&2 - p74-81 and 82-87	0.14	0.0015	200		Rainbow trout		mortality		32 days	dose calculated from reported dietary concentration and feeding rate; reported dose = 70 mg Cu/kg fish/32 d = 15.9 mg/kg bw/d.
Copper	copper sulphate pentahydrate	6.8		Lanno et al. 1985b	0.014	0.00034	287		Rainbow trout		growth		8 wks	feed refusal and 19% mortality was observed at 1,585 mg/kg-diet; feeding rate calculated from feed:gain ratio.
Copper	copper sulfate	13		Miller et al. 1993	0.016	0.00030	684		Rainbow trout		growth		42 days	feeding rate not reported; assumed average rainbow trout feeding rate of 1.9% bw/ day
Copper	Cu sulfate pentahydrate	14		Lundebye et al. 1999	0.0045	0.000126	500		Atlantic salmon	fry	growth		3 mos	Feeding rate not reported; assumed average Atlantic salmon feeding rate of 2.4% bw/ day
Copper	copper sulphate pentahydrate	17		Berntssen et al. 1999b	0.072	0.0017	691.3		Atlantic salmon	parr	growth	length, weight, or condition factor	4 wks	no significant differences between the treated groups and control, control showed a significantly higher final weight than starting weight, and treated groups did not; only no-effect level concentration reported; fish fed fish meal
Copper	copper sulphate pentahydrate	17		Lanno et al. 1985b	0.013	0.00029	664		Rainbow trout				24 wks	significant effects at 16 weeks; at 24 weeks, growth effects no longer apparent; only no-effect level concentration reported; fish fed prepared diet.
Copper	copper sulphate pentahydrate		18	Lanno et al. 1985b	0.0075	0.00019		730	Rainbow trout		growth		8 wks	not statistically tested, feed refusal and 19% mortality was observed at 1,585 mg/kg-diet
Copper	copper sulphate pentahydrate	18		Lanno et al. 1985b	0.0075	0.00019	730		Rainbow trout		mortality		8 wks	study showed potential effects at higher concentrations- confounded by food avoidance; feed refusal and 19% mortality were observed at 1,585-mg/kg diet; only no-effect level concentration presented here. Feeding rate calculated from feed:gain ratio (1.4g food/g BW gainX7.4g gain/56 days=0.185g feed/d)
Copper	copper sulphate pentahydrate		18	Lanno et al. 1985a	0.0091	0.00021		796	Rainbow trout		growth		16 wks	fish were fed on control diet for 12 days after exposure period.
Copper	Cu sulfate pentahydrate		20	Lundebye et al. 1999	0.0041	0.0001134		700	Atlantic salmon	fry	growth		3 mos	feeding rate not reported; assumed average Atlantic salmon feeding rate of 2.4% bw/ day
Copper	copper sulphate	20	28	Berntssen et al. 1999a	0.0048	0.00015	638	868	Atlantic salmon		growth		3 months	
Copper	copper sulphate pentahydrate	42		Kamunde et al. 2001	53.5	2.1	1,042		Rainbow trout		mortality/ growth		28 days	non-significant growth inhibition at NOAEL
Copper	copper chloride	26	50	Mount et al. 1994	0.00073	0.000044	440	830	Rainbow trout	fry	Mortality	69% and 95% surv at LOAEL and NOAEL	60 days	Fish were exposed to copper, cadmium, lead, and zinc in water at 23.0, 0.97, 3.32, and 46.3 µg/L, respectively, at the same time as the dietary exposure; fish exposed to live Artemia; mortality reported at LOAEL was possibly a result of aqueous copper exposure, not dietary copper
Copper	copper chloride	60		Mount et al. 1994	0.00060	0.000036	1,000		Rainbow trout	fry	growth	body weight and length	60 days	Fish were exposed to copper, cadmium, lead, and zinc in water at 23.0, 0.97, 3.32, and 46.3 µg/L, respectively, at the same time as the dietary exposure; fish exposed to live Artemia
Copper	copper sulphate pentahydrate		60	Baker et al. 1998	0.016	0.0004		2,400	Grey mullet		growth		67 days	Effects associated with reduced feeding
Copper	$CuSO_4$	69		Handy 1993b	0.13	0.00091	10,000		Rainbow trout		mortality		28 days	
Lead	lead nitrate	12.6		Mount et al. 1994	0.000633	0.000038	210		Rainbow trout	fry	mortality, growth	100% survival, body weight and length	60 days	Fish were exposed to copper, cadmium, lead, and zinc in water at 23.0, 0.97, 3.32, and 46.3 µg/L, respectively, at the same time as the dietary exposure
Lead	**not specified**	**134**		**Goettl et al. 1976**	**0.091**	**0.00173**	**7,040**		**Rainbow trout**	**immature**	**growth**		**191 days**	**feeding rate not reported; assumed average rainbow trout feeding rate of 1.9% bw/ day**

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Table 2. Summary of Updated Fish Dietary TRV Review for Round 2 ERA

Analyte	Chemical Form	NOAEL (mg/kg BW/day)	LOAEL (mg/kg BW/day)	Source	Body weight (kg)	FIR (kg-food/day)	NOAEL exposure concentration (mg/kg dw)	LOAEL exposure concentration (mg/kg dw)	Test Species	Lifestage	Endpoint	Endpoint Effect	Exposure Duration	Notes
Mercury	methylmercury		0.0091	Webber and Haines 2003		0.00018		0.51	Golden shiner		behavior	shoal area after predator exposure	90 days	growth was significantly reduced at this LOAEL; however, growth effect attributed to low food consumption in one replicate and growth effect was not dose-responsive (no effect in fish fed 0.959 ug/g ww). Initial weight calculated from data: avg of 0.8g of fish per liter, 20 fish per tank, each tank was 227 liters; doses are overestimated because 42% of feedings left in low-Hg diet tank vs 9-24% in other five tanks
Mercury	methylmercury		0.0192	Webber and Haines 2003		0.00018		1.1	Golden shiner		behavior	shoal area, shoal depth, and time to reform shoal after predator exposure	90 days	Initial weight calculated from data: avg of 0.8g of fish per liter, 20 fish per tank, each tank was 227 liters; doses are overestimated because 42% of feedings left in low-Hg diet tank vs 9-24% in other five tanks
Mercury	methylmercury	0.0192		Webber and Haines 2003		0.00018	0.959		Golden shiner		growth, mortality	growth, mortality	90 days	growth was significantly reduced at this LOAEL; however, growth effect attributed to low food consumption in one replicate and growth effect was not dose-responsive (no effect in fish fed 0.959 ug/g ww). Initial weight calculated from data: avg of 0.8g of fish per liter, 20 fish per tank, each tank was 227 liters; doses are overestimated because 42% of feedings left in low-Hg diet tank vs 9-24% in other five tanks
Mercury	methylmercury	0.0036	0.026	Friedmann et al. 1996	0.028	0.00073	0.69	4.9	Walleye	juvenile	growth; gonadal developme nt	male growth; male gonadosomatic index	6 months	fish fed 1.0 mg/kg mercury in diet consisting of catfish fillets injected with methylmercury 3X per week augmented twice monthly with mercury injected fathead minnows; no significant effect on mortality observed; no replication, elevated control mortality (28%), increased growth and body length in fish fed 0.1 mg/kg mercury in diet; no NOAEL for reproduction or growth endpoint because results from low and high doses were pooled; Note that NOAEL dose was indicated as LOAEL in the PRE.
Mercury	**methyl-mercuric chloride**	**0.013**	**0.048**	**Matta et al. 2001**	**0.011**	**0.000275**	**0.5**	**1.9**	**Mummic hog**		**mortality**	**F0 male survival**	**> = 6 wks**	**LOAEL previously reported as NOAEL in PRE; feeding rate not reported; assumed mummichog feeding rate from Gutjahr-Gobell et al. 1999**
Mercury	methyl-mercuric chloride		0.048	Matta et al. 2001	0.012	0.0003		1.9	Mummich og		mortality	F0 male survival	> = 6 wks	survival of female mummichog was unaffected at all doses; feeding rate not reported; assumed mummichog feeding rate from Gutjahr-Gobell et al. 1999
Mercury	methylmercury		0.053	Hammerschmidt et al. 2002	3.2	0.192		0.88	Fathead minnow	juvenile and adult	reproduc-tion	spawning success	multi-generation	spawning success at LOAEL = 46%; 75% in control; no statistics; feeding rate not reported; assumed fathead minnow feeding rate from Ogle and Knight 1989
Mercury	methyl-mercuric chloride	0.140		Matta et al. 2001	0.0131	0.0003275	5.6		Mummich og		reproduc-tion	F1 fertilization success	> = 6 wks	feeding rate not reported; assumed mummichog feeding rate from Gutjahr-Gobell et al. 1999
Mercury	methyl-mercury		0.46	Rodgers and Beamish 1982	0.013	0.00025		23.2	Rainbow trout	fingerling	growth	body weight in hatchling fish	12 weeks	growth was significantly reduced in fish fed 23.9 µg/g in diet ad libitum and in fish fed 23.3 µg/g in diet at a rate of 2% of their body weight (NOTE: no effect on growth was reported in fish fed 23.3 µg/g or 75.2 µg/g in diet at a rate of 1% of their body weight); food pellets were freeze dried.
Mercury	methyl-mercuric chloride	0.14	1.4	Matta et al. 2001	0.114	0.00285	5.6	54	Mummich og		reproduc-tion	F1 fertilization success	> = 6 wks	feeding rate not reported; assumed mummichog feeding rate from Gutjahr-Gobell et al. 1999

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Table 2. Summary of Updated Fish Dietary TRV Review for Round 2 ERA

Analyte	Chemical Form	NOAEL (mg/kg BW/day)	LOAEL (mg/kg BW/day)	Source	Body weight (kg)	FIR (kg-food/day)	NOAEL exposure concentration (mg/kg dw)	LOAEL exposure concentration (mg/kg dw)	Test Species	Lifestage	Endpoint	Endpoint Effect	Exposure Duration	Notes
Mercury	methyl-mercuric chloride	1.4		Matta et al. 2001	0.114	0.00285	54		Mummichog		reproduc-tion	F1 hatchability, survival, fecundity, F2 larval survival; fecundity, fertilization success, offspring weight, female survival	> = 6 wks	Offspring of medium and high doses were larger than offspring of control fish; feeding rate not reported; assumed mummichog feeding rate from Gutjahr-Gobell et al. 1999
Mercury	methyl-mercury	1.5		Rodgers and Beamish 1982	0.0138	0.0002208	94.8		Rainbow trout	fingerling	mortality		12 weeks	no mortality reported
Mercury	methyl-mercury		1.5	Rodgers and Beamish 1982	0.012	0.00024		76.5	Rainbow trout	fingerling	behavior	lethargic behavior and darker appearance of fish	12 weeks	behavioral effects were qualitatively observed in fish fed 94.8 µg/g ww in diet ad libitum and in fish fed 75.2 µg/g ww in diet at a rate of 2% of their body weight; food pellets were freeze dried.
Selenium	**seleno-L-methionine**	**0.10**	**0.20**	**Cleveland et al. 1993**	**0.0002**	**0.000006**	3.3	6.5	**Bluegill juveniles**	**juvenile**	**mortality**		**90 days**	
Selenium			0.24	Coughlan and Velte 1989	0.221	0.0046		9.6	Striped bass	juvenile	growth, mortality	condition factor; body weight; survival	up to 80 days	food avoidance observed; many more golden shiners consumed per feeding than red shiners; 0.3 ppm ww Se in WB (background); % moisture was approx. 75%. dose reported as average of 103 and 140 µg/fish/feeding x 3feedings/wk x fish/.221kg*mg/1000µg* 1 week/7 days
Selenium	seleno-L-methionine	0.2	0.4	Cleveland et al. 1993	0.0002	0.000006	6.5	13	Bluegill juveniles	juvenile	growth	condition factor	90 days	
Selenium	seleno-L-methionine; Na2SeO4; Na2SeO3	0.9	1.2	Ogle and Knight 1989	0.000605	0.000036	15	20	Fathead minnow	adult	growth		98 days + spawning periods	decreased palatability observed, food aversion- not strong effects
Selenium	seleno-L-methionine; Na2SeO4; Na2SeO3	1.8		Ogle and Knight 1989	0.000525	0.000032	30		Fathead minnow	adult	reproduc-tion		98 days + spawning periods	
Silver	**silver sulfide**	**70**		**Galvez and Wood 1999**	**0.0025**	**0.00005825**	**3000**		**Rainbow trout**		**growth**		**58 days**	
Zinc		**19.0**	**38.0**	**Takeda and Shimma 1977**	**0.01375**	**0.00026**	**1,000**	**2,000**	**Rainbow trout**	**fingerling**	**growth**		**6 wks**	**fish fed at same dose Zn with 0.5% Ca experienced no adverse effects; feeding rate not reported; assumed average rainbow trout feeding rate of 1.9% bw/ day**
Zinc	zinc chloride	114.0		Mount et al. 1994	0.00069	0.000041	1,900		Rainbow trout	fry	growth, mortality	body weight and length; 97% survival	60 days	Fish were exposed to copper, cadmium, lead, and zinc in water at 23.0, 0.97, 3.32, and 46.3 µg/L, respectively, at the same time as the dietary exposure
TBT	**tributyltin oxide**	**0.0021**	**0.020**	**Shimasaki et al. 2003**	**0.10342**	**0.0020684**	**0.106**	**1.02**	**Japanese flounder**	**larvae**	**growth, sex reversal**	**body weight, increased sex reversal to male**	**approx 65 days**	**all test fish were genetically XX fish by parents both XX but one parent phenotypically male and functional; growth effect significant at 100 days and was recovered after 300 days past hatching (growth not significantly affected at 300 days); no replication; survival not significantly affected, however, high mortality in all groups including control. Note that NOAEL and LOAEL changed. Recovered growth effect no longer considered significant; feeding rate not reported; assumed feeding rate of 2% bw/ day**

BZTO104(e)031228

Table 2. Summary of Updated Fish Dietary TRV Review for Round 2 ERA

Analyte	Chemical Form	NOAEL (mg/kg BW/day)	LOAEL (mg/kg BW/day)	Source	Body weight (kg)	FIR (kg-food/day)	NOAEL exposure concentration (mg/kg dw)	LOAEL exposure concentration (mg/kg dw)	Test Species	Lifestage	Endpoint	Endpoint Effect	Exposure Duration	Notes
PAHs (BaP)	**benzo(a)pyrene**	**0.66**		**Rice et al. 2000**		**0.0024**	**47**		**English sole**	**juvenile**	**growth**		**10-12 days**	**diet of exposed polychaetes exposed to contaminated sediment for 28 days; field collected fish. Growth rate at LOAEL concentration was 0.4%. Weight estimated from length using the equation y (g) = 0.7714x (mm) (Windward unpublished data); dose calculated from ww feeding rate and dietary concentration assuming 80% moisture content in prey.**
PAHs (BaP)	**benzo(a)pyrene**		**1.4**	**Rice et al. 2000**		**0.0020**		**116**	**English sole**	**juvenile**	**growth**		**10-12 days**	**diet of exposed polychaetes exposed to contaminated sediment for 28 days; field collected fish. Growth rate at LOAEL concentration was 0.4%. Weight estimated from length using the equation y (g) = 0.7714x (mm) (Windward unpublished data); dose calculated from ww feeding rate and dietary concentration assuming 80% moisture content in prey.**
PAHs (BaP)	benzo(a)pyrene		3.7	Hendricks et al. 1985	0.18			1,000	Rainbow trout	3 mo. old	growth	reduced body weight	18 months	Casein diet; total mortality was higher in the control group
PAHs (BaP)	benzo(a)pyrene	3.7		Hendricks et al. 1985	0.18		1,000		Rainbow trout	juvenile (3 mos old)	mortality		18 months	mortality was greater in control; NOAEL supported by additional 9-week study
PAHs (BaP)	benzo(a)pyrene	13		Wu et al. 2003		0.00133	81		areolated grouper	juvenile		survival, body length & weight	4 Wks + 4 wks recovery	exposure was dietary force feeding - fish were force fed pellet into larynx. Fish were force-fed 5% of their body weight per day. Concentration reported as 107.4 µg BaP/pellet. And a pellet was 5% of initial BW (2.66g) resulting in a concentration of 81 mg/kg.
PAHs (BaP)	benzo(a)pyrene	1.9	19	Hart and Heddle 1991	0.003	0.00006	100	1,000	Rainbow trout	juvenile	growth		28 days	feeding rate not reported; assumed average rainbow trout feeding rate of 1.9% bw/ day
PAHs (mixture)	Chemicals present in field collected sediments		0.12	Rice et al. 2000		0.0018		11.3	English sole	juvenile	growth		10-12 days	diet of exposed polychaetes exposed to contaminated sediment for 28 days; field collected fish. Growth rate at LOAEL concentration was 0.4%. Weight estimated from length using the equation y (g) = 0.7714x (mm) (Windward unpublished data)
PAHs (mixture)	PAH mixture[a]	5.0		Palm et al. 2003		0.00037		252	chinook salmon	juvenile	growth		7 wks	fish fed pelleted food with 10% moisture content
PAHs (mixture)	PAH mixture[a]	5.0		Palm et al. 2003		0.00037		252	chinook salmon	juvenile	disease resistance		4 wks exposure, 2 wks immuno-challenge	fish were exposed to Listonella anguillarum following PAH exposure. No difference was observed between PAH-exposed fish and controls in either fish that were vaccinated against the bacterium or those that weren't vaccinated; fish fed pelleted food with 10% moisture content
PAHs (mixture)	**PAH mixture[b]**	**6.1**	**18**	**Meador et al. 2006**		**0.00025**	**951**	**324**	**chinook salmon**	**smolt**	**growth**	**dry weight body weight**	**58 days**	**Lipid classes and Plasma chemistry were adversely affected at the NOAEL dose.**
PCBs	**Aroclor 1260**		**0.050**	**Hugla and Thome 1999**	**1**	**0.02**	**0.25**	**2.5**	**Barbel**	**adult**	**reproduct-ion**	**reduced fecundity, EROD induction**	**50 days**	**effects observed at lowest dose, therefore is an unbounded LOAEL; weight not reported; assumed typical adult barbel weight of 1kg; feeding rate not reported; assumed feeding rate of 2% bw/ day**
PCBs	Aroclor 1260		0.25	Hugla and Thome 1999	1	0.02		12.5	Barbel	adult	reproductio n	failure to spawn	50 days	weight not reported assumed typical adult barbel body weight of 1 kg; feeding rate not reported; assumed feeding rate of 2% bw/ day
PCBs	Aroclor 1254	0.34		Powell et al. 2003	11.86	0.24	18		Chinook salmon	juvenile	growth			whole body burdens ranged from 0.74 to 0.98 mg/kg ww over the 13-day period following treatment; fish fed pelleted salmon feed - 7.5% moisture; dose calculated from ww dietary concentration and ww feeding rate

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Table 2. Summary of Updated Fish Dietary TRV Review for Round 2 ERA

Analyte	Chemical Form	NOAEL (mg/kg BW/day)	LOAEL (mg/kg BW/day)	Source	Body weight (kg)	FIR (kg-food/day)	NOAEL exposure concentration (mg/kg dw)	LOAEL exposure concentration (mg/kg dw)	Test Species	Lifestage	Endpoint	Endpoint Effect	Exposure Duration	Notes
PCBs	Aroclor 1254		0.40	McCarthy et al. 2003	0.4				Atlantic croaker	adult	reproduc-tion	impaired larval response to startle stimulus		parental fish fed dietary PCBs; fish fed contaminated ground shrimp; dietary concentration not reported but dietary dose was reported
PCBs	Aroclor 1254		3.80	Hendricks et al. 1981	0.36	0.00684		200	Rainbow trout	adult	reproduc-tion	offspring fry growth		gravid females fed 200 µg/g PCBs for 60 days; feeding rate not reported; assumed average rainbow trout feeding rate of 1.9% bw/ day
PCBs	Aroclor 1254		0.018	Berlin et al. 1981	1.9	0.0331245		1	Brook trout	fry	mortality	fry mortality	176 days	field-collected eggs from Lake Michigan with starting egg residues of 7.6 µg/g PCBs and 4.7 µg/g DDE; mortality is estimated; dose calculated from ww dietary concentration assuming the average brook trout feeding rate of 1.7% bw/ day
PCBs	Aroclor 1260	0.48		Mayer et al. 1977	0		24		Channel catfish	fingerling	growth, mortality			food type not reported; fish force fed PCBs in capsules (gavage)
PCBs	Aroclor 1242		0.6	Hansen et al. 1976	0.0215	0.00065		20	Channel catfish	fingerling	growth	reduced weight gain (60% of controls)	20 wks	
PCBs	Aroclor 1254	0.75		Lieb et al. 1974	0.039		15		Rainbow trout	14-wk	growth, mortality			
PCBs	Aroclor 1268	0.800		Matta et al. 2001	0.124	0.0031	32		Mummichog	adult	reproduc-tion, growth, survival	fertilization and hatching success, larval survival, weight, survival	> = 6 wks	two generations of progeny observed; only no-effect level reported; feeding rate not reported; assumed mummichog feeding rate from Gutjahr-Gobell et al. 1999
PCBs	Aroclor 1254	1.5	14.5	Mayer et al. 1977	NR		48	480	Coho salmon	fingerling	mortality			food type not reported; all fish died within 265 days of dose in LOAEL group (no reported mortality in NOAEL group); no statistics
DDT	p,p'-DDT	0.0023		Warlen et al. 1977	0.35	0.00017	0.093		Atlantic menhaden		growth	growth	48 days	
DDT	p,p'-DDE		0.0044	Berlin et al. 1981	1.8	0.0299625		0.28	Brook trout	fry	mortality	fry mortality	176 days	field-collected eggs from Lake Michigan with starting egg residues of 7.6 µg/g PCBs and 4.7 µg/g DDE; mortality is estimated; dose calculated from ww dietary concentration assuming the average brook trout feeding rate of 1.7% bw/ day
DDT	sum of detected DDT and its metabolites (DDD and DDE)	0.019		Rhead and Perkins 1984	0.00455	0.000091	0.93		Goldfish	9.1 g	mortality	survival	58 days	feeding rate not reported; assumed feeding rate of 2% bw/ day
DDT	total DDT	0.036		Hilton et al. 1983	0.068	0.00129	1.89		rainbow trout	juvenile	growth, mortality, behavior	final body weight, survival, abnormal behavior	24 wks	feeding rate not reported; assumed average rainbow trout feeding rate of 1.9% bw/ day
DDT	sum of detected DDT and its metabolites (DDD and DDE)	0.045		Macek and Korn 1970	0.12	0.0018	3.0		Brook trout	juvenile	mortality		120 days	no effect level reported; fish fed 3.0 mg/kg dw once daily at a rate of 1.5% of their body weight/day; fish fed pellet diet; length weight relationship based on (http://wlapwww.gov.bc.ca/nor/fish/stocking/ferguson/)

BZTO104(e)031232

Table 2. Summary of Updated Fish Dietary TRV Review for Round 2 ERA

Analyte	Chemical Form	NOAEL (mg/kg BW/day)	LOAEL (mg/kg BW/day)	Source	Body weight (kg)	FIR (kg-food/day)	NOAEL exposure concentration (mg/kg dw)	LOAEL exposure concentration (mg/kg dw)	Test Species	Lifestage	Endpoint	Endpoint Effect	Exposure Duration	Notes
DDT	sum of detected DDT and its metabolites (DDD and DDE)	0.14		Macek et al. 1970	0.055	0.0011	1.0		Rainbow trout	15 g	growth, mortality	survival, growth	168 days	
DDT	**total DDT**		**0.14**	**Macek 1968 (j. fish res. Brd Ca 25(9)**	**0.081**	**0.0012**		**7.2**	**Brook trout**	**sex mat yearlings**	**reproduc-tion**		**156 days**	**embryo mortality; LOAEL= reproduction effect in fish fed 1.0 mg/kg/wk (or 7.2 mg/kg DDT in diet)**
DDT	technical grade DDT	0.3		Macek 1968a	0.2725			13.0	Brook trout fry	fry	growth, mortality	survival, growth	31 weeks	other fish species from study not reported because NOAEL/LOAELs are much higher; length but not weight of male fish was significantly greater than controls; NOAEL- no effect on growth or mortality in fish fed 2.0 mg/kg/wk (or 14.4 mg/kg DDT in diet)
DDT	sum of DDD, DDE, and DDT		0.29	Macek 1968b (j. fish res. Brd Ca 25(11)	0.037	0.00067		13	Brook trout	under yearling	mortality	reduced survival during stress (starvation)	26 wks	LOAEL is range of total DDT (including DDD, DDE, 2,4'-DDT and 4,4'-DDT) in fish exposed to 2.0 mgkg/wk DDT; controls found to have 0.36 µg/kg DDT (0.0071 mg/kg/d).
DDT	sum of DDD, DDE, and DDT		0.29	Macek 1968b (j. fish res. Brd Ca 25(11)	0.036	0.00067		13	Brook trout	under yearling	growth	increased growth decreased survival during starvation	31 wks	growth LOAEL effect is an increase in growth (significant) and not shown to be an adverse effect; controls found to have 0.36 µg/kg DDT (0.0071 mg/kg/d). Survival LOAEL is during starvation.
DDT	"DDT" including DDE and DDD		0.81	Butler et al. 1969	0.003	0.00060		4	pinfish	4.8 g	mortality	44% mortality at 10 days	15 days	4% mortality in control; dose calculated from ww dietary concentration and ww feeding rate
DDT	sum of DDE and DDD detected in tissue	0.48	1.16	Buhler et al. 1969	0.000725	0.000022475	19	75	Chinook salmon	0.61 g	mortality	survival	up to 95 days	growth increase attributed to size-selective mortality (smaller fish were more susceptible to DDT toxicity); dose calculated from ww dietary concentration and ww feeding rate
DDT	"DDT" including DDE and DDD		1.4	Butler et al. 1969	0.0048	0.00144		5	pinfish	3.0 g	mortality	35% mortality at 21 days	21 days	0% mortality in control; dose calculated from ww dietary concentration and ww feeding rate
DDT	sum of detected DDT and its metabolites (DDD and DDE)	0.63	2.50	Buhler et al. 1969	0.004615	0.000115375	50	200	Coho salmon	3.7 g	mortality	survival	up to 95 days	dose calculated from ww dietary concentration and ww feeding rate
DDT	sum of detected DDT and its metabolites (DDD and DDE)		2.7	Jarvinen et al. 1976, 1977		0.0000072		45.6	Fathead minnow	juvenile-adult	mortality	survival	266 days	Probability of survival in exposed fish was approximately 78% versus approximately 90% survival in controls, the difference was statistically significant; fish fed dried clam tissue diet; Note previously noted as a NOAEL; feeding rate not reported; dose calculated assuming average fathead minnow feeding rate of 6% bw/ day
DDT	sum of detected DDT and its metabolites (DDD and DDE)	0.75	3.00	Buhler et al. 1969	0.00181	0.0000543	50	200	Chinook salmon	1.1 g	mortality	survival	up to 95 days	dose calculated from ww dietary concentration and ww feeding rate

[a] PAH mixture includes: acenaphthene, benz[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene, benzo[ghi]perylene, benzo[k]fluoranthene, chrysene, dibenz[a,h]anthracene, fluoranthene, fluorene, indeno[1,2,3-cd]pyrene, naphthalene, phenanthrene, pyrene
b PAH mixture includes:naphthalene, 2-methylnaphthalene, dimethylnaphthalene, dibenzothiophene, acenaphthene, fluorene, 1,8-dimethyl(9H)fluorene, phenanthrene, 9-ethylphenanthrene, 9-ethyl-10-methylphenanthrene, 1-methyl-7-isopropylphenanthrene, anthracene, fluoranthene, pyrene, methyl pyrene, ben

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Table 3. Summary of Updated Bird Dietary TRV Review for Round 2 ERA

Analyte	NOAEL (mg/kg bw/d)	LOAEL (mg/kg bw/d)	Source	Endpoint	Test Species	Chemical Form	Exposure Mode	FI (kg dw or L/day)	Wet or Dry?	FI Default?	Nagy bird guild	Body Weight (kg)	BW Default?	% Moisture	NEC wet (ppm)	NEC dry (ppm)	LEC wet (ppm)	LEC dry (ppm)	Exposure Duration	Effect Endpoint	Notes
PCBs (Aroclor 1254)	0.054		Ahmed et al. 1978	mortality, growth, reproduction	White leghorn males	Aroclor 1254	food	0.0034	W			2.56			40				20 wks	fertility; hatchability, growth, mortality	no control values given
PCBs (Aroclor 1248)		0.35	Lowe and Stendell 1991	reproduction	American kestrel	Aroclor 1248	food	0.0136	D	1	6	0.13	I	10%			3	3.3	5.5 months	eggshell weight and thickness	only one dose used
PCBs (Aroclor 1248)	0.49		McLane and Hughes 1980	reproduction	Screech owl	Aroclor 1248	food	0.0266	D	1	5	0.181	B	10%	3	3.3333			2 generations	eggshell thickness, egg production, hatching success, fledging success	egg tissue concentrations also reported in study
PCBs (Aroclor 1242)	**0.29**	**0.58**	**Britton and Huston 1973**	**reproduction**	**White leghorn chickens**	**Aroclor 1242**	**food**	**0.0997**	**W**	**3**		**1.71**	**C**		**5**		**10**		**6 wks + 5 wks untreated**	**hatchability**	**significant effects on hatchability**
PCBs (Aroclor 1242)		0.60	Hill et al. 1975a	reproduction	Japanese quail	Aroclor 1242	food	0.0048	D	1	3	0.09	B	10%			10	11.111	45 days	eggshell thinning	only one dose used
PCBs (Aroclor 1248)	0.061	0.61	Scott et al. 1975	reproduction	White leghorn chickens	Aroclor 1248	food	0.105	W			1.71	C		1		10		8 wks	egg production and egg hatchability	egg residues also reported
PCBs (Aroclor 1232)		1.2	Cecil et al. 1974	reproduction	White leghorn hens	Aroclor 1232	food	0.0997	W	3		1.71	C				20		9 wks + 7 wks untreated then mated	hatchability, embryo abnormality, embryo mortality	only one dose used; no discussion of statistical significance
PCBs (Aroclor 1254)		1.4	Peakall et al. 1972; Peakall and Peakall 1973	reproduction	Ringed turtle-dove	Aroclor 1254	food	0.0202	D	1	1	0.155	E	9%			10	10.989	2 generations	hatching success in second generation	egg tissue concentrations also reported in study
PCBs (Aroclor 1254)		1.6	Dahlgren et al. 1972	reproduction	Ring-necked pheasant	Aroclor 1254	gelatin capsule					1.135	B				1.7857		Once per wk for 16 wks	egg hatchability	dose reported in mg/kg/wk; daily dose derived from weekly dose (7 mg/week/7)
PCBs (Aroclor 1254)		1.6	Tori and Peterle 1983	behavior	Mourning dove	Aroclor 1254	food	0.0168	D	1	1	0.119	B	10%			10	11.1	42 days (+30 days untreated following 2 wks post exposure)	reduced courtship behavior, fewer successful pair bonds formed (both statistically significant); also delay onset of nest initiation	unbounded LOAEL
PCBs (Aroclor 1254)	2.5		Custer and Heinz 1980	reproduction	Mallard	Aroclor 1254	food	0.1082	W	2		1.082	B		25				~ 1 month	Reproductive success	
PCBs (Aroclor 1254)	0.29	2.9	Platonow and Reinhart 1973	reproduction	White leghorn chickens	Aroclor 1254	food	0.0997	W	3		1.71	C		5		50	50.0	39 wks (14 wks for 50 ppm group)	hatchability	statistically significant effect observed; LOAEL is residues where instantaneous depression of hatchability and embryotoxicity is observed; NOAEL is where hatchability of fertile eggs is unaffected; however, at NOAEL fertility and egg production are significantly reduced (study attributes it to mating inactivity and not PCB exposure)- uncertain NOAEL
PCBs (Aroclor 1254)	3.9		Risebrough and Anderson 1975	reproduction	Mallard	Aroclor 1254	food	0.1082	W	2		1.082	B		39				4 months	egg production, eggshell thinning	
PCBs (Aroclor 1248: 1254:1260 mixture)		5 to 7	Fernie et al. 2000, 2001	reproduction	American kestrel	1:1:1 ratio of Aroclor 1248:1254:1260	food												100 days until eggs hatched	egg laying in second generation (exposed in ovo); also some effect on clutch size and fledgling success	body weight-normalized dose estimated in study; no statistics- egg laying endpoint: 91% in controls laid a clutch of eggs; 75% in test group
PCBs (Aroclor 1248: 1254:1260 mixture)		5 to 7	Fisher et al. 2001	reproduction	American kestrel	1:1:1 ratio of Aroclor 1248:1254:1260	food												1 mo prior to mating through mating period	courtship behavior	body weight normalized dose estimated in study; no adverse effect on male sexual behavior and no change in female sexual behavior or frequency of copulation; study performed along with Fernie et al. 2000; 2001
PCBs (Aroclor 1248: 1254:1260 mixture)		5 to 7	Fernie et al. 2003	reproduction	American kestrel	1:1:1 ratio of Aroclor 1248:1254:1260	food												1 month prior to pairing until anticipated egg hatching	cracked eggs; embryo abnormalities;	birds were fed injected cockerels
PCBs (Aroclor 1248: 1254:1260 mixture)		7	Fernie et al. 2003a	reproduction	American kestrel	1:1:1 ratio of Aroclor 1248:1254:1260	food										4.85		100 days	offspring mortality and offspring body weight	birds were fed injected cockerels, stats not presented but 52% of offspring died within 3 days of hatching; no adverse effect observed in offspring body weight (actually significantly increased at this LOAEL dose for offspring mortality; no stats provided

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Table 3. Summary of Updated Bird Dietary TRV Review for Round 2 ERA

Analyte	NOAEL (mg/kg bw/d)	LOAEL (mg/kg bw/d)	Source	Endpoint	Test Species	Chemical Form	Exposure Mode	FI (kg dw or L/day)	Wet or Dry?	FI Default?	Nagy bird guild	Body Weight (kg)	BW Default?	% Moisture	NEC wet (ppm)	NEC dry (ppm)	LEC wet (ppm)	LEC dry (ppm)	Exposure Duration	Effect Endpoint	Notes
PCBs (Aroclor 1248 1254:1260 mixture)		5 to 7	Fernie et al. 2003b	reproduction	American kestrel	1:1:1 ratio of Aroclor 1248:1254:1260	food												100 days	offspring growth rate reduced	birds were fed injected cockerels; unclear whether reduced growth rate in F2 nestlings was statistically evaluated; however, other studies indicate that effects in in F1 generation and parents were statistically significant at this dose
PCBs (Aroclor 1254)		9.5	Bird et al. 1983	reproduction	American kestrel	Aroclor 1254	food										33		62-69 days	decreased sperm count and sperm concentration	endpoint is not a direct measure of reproductive success; assumed 80% moisture from day-old dead chicks in diet
PCBs (Aroclor 1254)		12.0	Kreitzer and Heinz 1974	behavior	Japanese quail	Aroclor 1254	food	0.0048	D	1	3	0.09	B	10%			200	222.2	8 days treated + 6 days untreated	avoidance response (depressed response to stimuli)	statistically significant effect; only one dose used
PCBs (Aroclor 1242)		15	Haseltine and Prouty 1980	reproduction	Mallard	Aroclor 1242	food	0.1082	W	2		1.082	B				150		12 wks	hatchability, embryo mortality, egg viability, embryo abnormalities	egg tissue concentrations also reported in study
PCBs (Aroclor 1254)	NC	NC	Stickel et al. 1984	mortality	common grackles, red-winged blackbirds, brown-headed cowbird, starling	Aroclor 1254	food											1500	birds fed until 50% of birds died	study not useful- presents LT50 in four bird species at an extremely high dietary PCB concentration	
Cadmium		0.40	Heinz and Haseltine 1983	behavior	Black duck	cadmium chloride	food	0.1250	W	2		1.25	B	10%			4		4 months prior to egg laying- 1 wk after hatching	avoidance response in 7 day old chicks (distance run from fright stimulus)	both parents and chicks (until 1 wk of age) were fed Cd diet; avoidance response was significantly altered at LOAEL but not at 40 ppm (no clear dose-response); study cites that "failure to respond normally to predators might result in increased mortality"
Cadmium	1.6		Cain et al. 1983	growth	Mallard-young	cadmium chloride	food	0.0883	D	2		0.8825		10%	14.6	16.222			12 wks	body weight, femur density	female ducks
Cadmium	**0.73**	**2.9**	**Leach et al. 1979**	**reproduction**	**Leghorn hen**	**$CdSO_4$*$8H_2O$**	**food**	**0.1019**	**W**	**3**		**1.70**	**C**		**12.22**		**48.22**		**48 wks**	**egg production, shell thickness**	
Cadmium		4.0	Richardson et al. 1974	growth	Japanese quail (chicks)	cadmium chloride	food	0.0050	D	1	3	0.093	G					75	6 wks	male body weight	
Cadmium	19		White and Finley 1978b	mortality	Mallard	cadmium chloride	food	0.110	W			1.153			200				90 days		focus on tissue accumulation; no statistics; 0% mortality
Cadmium	20		White and Finley 1978a	mortality, growth	Mallard	cadmium chloride	food	0.110	W			1.153			210				30-90 days	body weight, adult mortality	no statistics for mortality; 0% mortality
Cadmium		24	Freeland and Cousins 1973	growth	Leghorn chicks	cadmium chloride	food	0.0434	D	4		0.138						75	21 days	male body weight	
Cadmium	1.5	20	White and Finley 1978a	reproduction	Mallard	cadmium chloride	food	0.110	W			1.153			15.2		210		30-90 days	egg production	effects not consistent w/ dose; statistics not presented; overall variable number of hens laid eggs
Cadmium		40	Pintzi et al. 1974	growth	Leghorn chicks	cadmium chloride	food	0.0577	D	4		0.574	A					400	20 days	male body weight	
Cadmium	16		DiGiulio and Scanlon 1984	growth	Mallard	cadmium chloride	food	0.122	W			1.119			150				42 day	body weight	
Cadmium		47	DiGiulio and Scanlon 1984	growth	Mallard	cadmium chloride	food	0.108	W			1.027					450		42 day	body weight	statistically significant effects at LOAEL
Chromium	0.10		Lien et al. 2004	reproduction	White leghorn layers (hens)	chromium picolinate	food	0.1019	W	3		1.70	C		1.6				28 days	egg weight and shell thickness	Effect of eggshell thinning on reproductive success (e.g., hatchability or offspring viability) was not reported
Chromium	0.17		Jensen and Maurice 1980	reproduction	leghorn laying hen	$CrCl_3$ * H_2O	food	0.0997	W	3		1.71	C		2.9484				4 wks	egg production and wt, interior quality	Cr+3
Chromium	**1.0**	**5.0**	**Haseltine et al. unpub as cited in Sample 1996**	**reproduction**	**Black duck**	**Cr 3+ as $CrK(SO_4)_2$**	**food**	**0.1250**	**W**	**2**		**1.25**	**B**		**10**		**50**		**10 months (and critical lifestage)**	**duckling survival**	**Original paper could not be located**
Chromium	7.7		Romoser et al. 1961	mortality, growth	Nichols chicks	Na2CrO4	food	0.0286	W	4		0.12	A		32.138				22 days	male adult mortality, male body weight	
Chromium		22.8	Burger and Gochfeld 1995b	growth, behavior	Herring gull (chicks)	Chromium nitrate	single ip injection												single injection + 48 days untreated	depressed growth and altered behavior- balance, depth perception, thermoregulation, and begging behavior	single intraperitoneal injection; behavior endpoints monitored until chicks were 50 days of age
Chromium		105	Chung et al. 1985	growth, mortality	broiler chicks	Cr6+ (fm K_2CrO_4)	food	0.0295	W	4		0.254	A				900		2 wks	body weight, adult mortality	Cr+6
Chromium		465	Chung et al. 1985	growth	broiler chicks	Cr6+ (fm $Cr_2(SO_4)_3$*$4H_2O$	food	0.0295	W	4		0.254	A				4000		2 wks	body weight	
Chromium	465		Chung et al. 1985	mortality	broiler chicks	Cr6+ (fm $Cr_2(SO_4)_3$*$4H_2O$	food	0.0295	W	4		0.254	A		4000				2 wks		unclear controls

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Table 3. Summary of Updated Bird Dietary TRV Review for Round 2 ERA

Analyte	NOAEL (mg/kg bw/d)	LOAEL (mg/kg bw/d)	Source	Endpoint	Test Species	Chemical Form	Exposure Mode	FI (kg dw or L/day)	Wet or Dry?	FI Default?	Nagy bird guild	Body Weight (kg)	BW Default?	% Moisture	NEC wet (ppm)	NEC dry (ppm)	LEC wet (ppm)	LEC dry (ppm)	Exposure Duration	Effect Endpoint	Notes
Copper	2.1		Dozier et al. 2003	growth, survival	Broiler chicks	Copper sulfate, copper amino acid complex	food	0.0295	D	4		0.254	A			18			17 days	body weight, mortality	unbounded NOAEL - NOAEL is backgound concentration (6 ppm) in diet plus exposure concentration (12 ppm)
Copper	11.2		Balevi & Coskun 2004	reproduction, survival	Hisex-Brown hens	Copper sulfate (hydrous)	food	0.09585	D			1.71	C	8.30%		200			90 days	damaged egg ratio, egg weight and survival	
Copper	15		Lien et al. 2004	reproduction	White leghorn layers (hens)	Copper sulfate	food	0.1019	W	3		1.70	C		250				28 days	egg weight and shell thickness	
Copper	16	29	Smith 1969	growth	Chicks, day-old	Copper sulfate	food	0.044	W	5		0.534	F		200		350		25 days	growth	
Copper	21	41	Poupoulis and Jensen 1976	growth	Chicks	Copper sulfate	food	0.044	W	5		0.534	F		250		500		4 wks	growth and gizzard erosion	
Copper	**47**	**62**	**Mehring et al. 1960**	**growth/ mortality**	**Chicks**	**Copper oxide**	**food**	**0.044**	**W**	**5**		**0.534**	**F**		**570**		**749**		**10 wks**	**growth, mortality**	
Copper		66	Persia et al. 2004	growth	Chicks	Copper chloride	food	0.0509	D			0.3845						500	8-22 days	body weight	selected lowest LOAELs observed over the 4 experiments
Lead		0.2	Edens and Garlich 1983; Edens et al. 1976	reproduction	Japanese quail, from day of hatching	Lead acetate	food	0.031	W	6		0.155	G				1		5 wks	egg production	
Lead	0.2	2.0	Edens and Garlich 1983	reproduction	Japanese quail, 6 weeks old	Lead acetate	food	0.031	W	6		0.155	G		1		10		5 wks	egg production	
Lead	2.5		Finley et al. 1976	mortality	Mallards, first-year	Lead nitrate	food	0.1082	W	2		1.082	B		25				12 wks	mortality, pathologic lesions	
Lead	1.64	3.3	Edens and Garlich 1983	reproduction	Leghorn chickens	Lead acetate	food	0.121	W	6		1.84	G		25		50		4 wks	egg production	
Lead	5.82		Pattee 1984	mortality/ reproduction	American kestrel	Metallic lead powder	food	0.0136	D	1	6	0.13		10%	50	55.556			5-7 months	mortality, fertility, egg production, eggshell thinning	
Lead	5.5	28	Morgan et al. 1975	growth	Japanese quail (chicks)	Lead acetate	food	0.0040	D	1	3	0.0715		10%		100		500	6 wks	body weight	
Lead	10.9		Kendall and Scanlon 1981	reproduction	Ringed Turtle-Dove	Lead acetate	Drinking water	0.017	W	8		0.155	E		100				2 wks prior to mating through breeding (11 wks)	onset of egg laying, egg production, juvenile body weight, egg fertility	only one dose used
Lead	**2.0**	**20**	**Edens et al. 1976**	**reproduction**	**Japanese quail**	**Lead acetate**	**food**	**0.031**	**W**	**6**		**0.155**	**G**		**10**		**100**		**12 wks**	**egg hatchability**	
Lead	75		Kendall and Scanlon 1982	growth	Ringed turtle-dove	Lead acetate	Oral intubation												7 days	Growth, ALAD activity	
Lead	25	125	Hoffman et al. 1985	growth	American kestrel nestlings	Metallic lead powder	Oral intubation												10 days	Growth	
Lead	50	100	Burger and Gochfeld 1995a	behavior	Herring gull (chicks)	Lead acetate	single ip injection												single injection + 42 days untreated	thermoregulation behavior (increased time to reach shade)	single intraperitoneal injection; behavior endpoint not direct measure of mortality, growth, or reproduction
Lead		100	Burger 1990	behavior	Herring gull (chicks)	Lead nitrate	single ip injection												single injection + 45 days untreated	altered behavior- righting response, balance, individual recognition, begging, thermoregulation, fish swallowing speed and depth perception	single intraperitoneal injection; unbounded LOAEL; endpoints monitored until day 45 (day-old chicks at start)
Lead		100	Burger and Gochfeld 1993	behavior	Herring gull (chicks)	Lead nitrate	single ip injection												single injection + 18 days untreated	altered behavior- response time, locomotor activity, required longer time to reach food	single intraperitoneal injection; endpoints monitored until day 19 (day-old chicks at start)
Lead		100	Burger and Gochfeld 1994	growth, behavior	Herring gull (chicks)	Lead acetate	single ip injection												single injection + 21 days untreated	depressed growth (temporarily significant) and altered behavior- depressed begging and walking activity, increased error rate in pecking behavior while feeding	single intraperitoneal injection
Lead		200	Gochfeld and Berger 1988	growth, behavior	Common tern (chicks)	Lead nitrate	single ip injection												single injection + 21 days untreated	depressed growth and reduced feeding efficiency (ability to manipulate fish)	single intraperitoneal injection
Lead		200	Burger and Gochfeld 1988	behavior	Common tern (chicks)	Lead nitrate	single ip injection												two injections at day 2 and 7	altered behavior- locomotion, righting response, and depth perception	endpoints monitored for 17 days; single intraperitoneal injection
Lead		200	Burger and Gochfeld 1995b	growth, behavior	Common tern (chicks)	Lead nitrate	single ip injection												single injection + 25 days untreated	depressed growth and altered behavior- locomotion, balance, righting response, feeding tasks, depth perception, and thermoregulation	single intraperitoneal injection
Lead		400	Burger and Gochfeld 1988	mortality	Common tern (chicks)	Lead nitrate	single ip injection												two injections at day 2 and 7	20% survival	endpoints monitored until day 17; single intraperitoneal injection; 100% survival in control birds
Selenium	0.025		Choct et al. 2004	growth, mortality	Broiler chicks	sodium selenite	food	0.1007	D			0.993	A				0.25		~40 days	body weight	
Selenium	0.025		Choct et al. 2004	growth, mortality	Broiler chicks	sel-plex 50	food	0.0982	D			0.993	A				0.25		~40 days	body weight	
Selenium		**0.82**	**Heinz et al. 1989**	**reproduction**	**Mallard**	**selenomethionine**	**food**	**0.1145**	**W**	**2**		**1.145**		**11%**			**8.15**		**~100 days**	**offspring growth/survival**	**statistically significant effects at LOAEL**
Selenium	**0.42**		**Heinz et al. 1989**	**reproduction**	**Mallard**	**selenomethionine**	**food**	**0.1158**	**W**	**2**		**1.158**		**11%**	**4.15**				**~100 days**	**offspring growth/survival**	**statistically evaluated**

Table 3. Summary of Updated Bird Dietary TRV Review for Round 2 ERA

Analyte	NOAEL (mg/kg bw/d)	LOAEL (mg/kg bw/d)	Source	Endpoint	Test Species	Chemical Form	Exposure Mode	FI (kg dw or L/day)	Wet or Dry?	FI Default?	Nagy bird guild	Body Weight (kg)	BW Default?	% Moisture	NEC wet (ppm)	NEC dry (ppm)	LEC wet (ppm)	LEC dry (ppm)	Exposure Duration	Effect Endpoint	Notes
Selenium		1.0	Heinz et al. 1987	reproduction	Mallard	Na_2SeO_3	food	0.105	W			1.046		9.5%			10		4 wks+laying(F0), 3 wks ducklings	embryo abnormalities	
Selenium	0.50		Heinz et al. 1987	reproduction	Mallard	Na_2SeO_3	food	0.104	W			1.036		9.5%	5				4 wks+laying(F0), 3 wks ducklings	embryo abnormalities	
Selenium	1.6		Heinz et al. 1989	mortality, growth	Mallard	selenomethionine	food	0.1107	W	2		1.107		11%	16.15				~100 d	body weight; adult mortality	no statistics for mortality; 0% mortality; growth effect may be result
Selenium		2.5	Heinz et al. 1987	growth	Mallard	Na_2SeO_3	food	0.094	W			0.938		9.5%			25		4 wks+laying(F0), 3 wks ducklings	adult growth	
Selenium	1.0		Heinz et al. 1987	growth	Mallard	Na_2SeO_3	food	0.105	W			1.046		9.5%	10				4 wks+laying(F0), 3 wks ducklings	adult growth	
Selenium		3.2	Wiemeyer and Hoffman 1996	growth	Screech owl	selenomethionine	food	0.050	W			0.189		11%			12.1	30	~ 3 mos	body weight	Food avoidance may have affected growth; calc NOAEL (Sample) 0.44; LOAEL = 1.5
Selenium	1.0		Wiemeyer and Hoffman 1996	growth	Screech owl	selenomethionine	food	0.050	W			0.184		11%	3.7	8.8			~ 3 mos	body weight	Food avoidance may have affected growth; calc NOAEL (Sample) 0.44; LOAEL = 1.5
Selenium		3.2	Wiemeyer and Hoffman 1996	reproduction	Screech owl	selenomethionine	food	0.050	W			0.189		11%			12.1	30	~ 3 mos	hatching success, 5 day survival, clutch size, egg size and mass	
Selenium	1.0		Wiemeyer and Hoffman 1996	reproduction	Screech owl	selenomethionine	food	0.050	W			0.184		11%	3.7	8.8			~ 3 mos	hatching success, 5 day survival, clutch size, egg size and mass	
Selenium	4.6		Heinz et al. 1988	mortality	Mallard	Na_2SeO_3	food	0.075	W			0.326		7%	20.1				42 days		selenomethionine also tested - less toxic
Selenium		4.6	Heinz et al. 1988	growth	Mallard	Na_2SeO_3	food	0.075	W			0.326		7%			20.1		42 days	body weight	Food avoidance observed; selenomethionine also tested - less
Selenium	2.1		Heinz et al. 1988	growth	Mallard	Na_2SeO_3	food	0.097	W			0.459		7%	10.1				42 days	body weight	Food avoidance observed. Selenomethionine also tested - less
Selenium		10	Heinz et al. 1987	mortality	Mallard	Na_2SeO_3	food	0.072	W			0.718		9.5%			100		4 wks+laying(F0), 3 wks ducklings	adult mortality	no statistics; 100% mortality at LOAEL; 1 male mortality at NOAEL
Selenium	2.5		Heinz et al. 1987	mortality	Mallard	Na_2SeO_3	food	0.094	W			0.938		9.5%	25				4 wks+laying(F0), 3 wks ducklings	adult mortality	no statistics; 100% mortality at LOAEL; 1 male mortality at NOAEL
Selenium		10	Heinz et al. 1988	mortality	Mallard	Na_2SeO_3	food	0.038	W			0.146		7%			40.1		42 days		selenomethionine also tested - less toxic
Zinc	17		Dozier et al. 2003	growth, mortality	Broiler chicks	Copper sulfate, copper amino acid complex	food	0.0295	D	4		0.254	A			150			17 days	body weight, mortality	NOAEL is background concentration (30 ppm) in diet plus exposure concentration (120 ppm)
Zinc	**82**	**124**	**Roberson and Schaible 1960**	**growth**	**White rock chicks**	**Zinc oxide, zinc sulfate, or zinc carbonate**	**food**	**0.044**	**W**	**5**		**0.534**	**F**		**1000**		**1500**		**5 wks**	**growth**	**domestic species**
Zinc		300	Gasaway and Buss 1972	mortality	Mallard (7 wk old)	Zinc carbonate	food	0.1082	W	2		1.082	B				3000		60 days	mortality, leg paralysis	
Zinc		344	Persia et al. 2004	growth	Chicks	Zinc chloride	food	0.0395	D			0.2873						2500	8-22 days	body weight	selected lowest LOAELs observed over the 4 experiments
Zinc	330	659	Oh et al. 1979	growth/ mortality	Hubbard broiler chicks	Zinc acetate	food	0.044	W	5		0.534	F		4000		8000		5 wks	mortality, reduced growth	
Zinc	133		Stahl et al. 1990	reproduction	White leghorn hens	Zinc sulfate	food and supplements	0.125	W			1.9			2028				44 wks	egg hatchability	
PAHs (mixture)	**40**		**Patton and Dieter 1980**	**growth**	**Mallard**	**aromatic hydrocarbon mixture including individual PAHs**	**food**	**0.1233**	**W**	**2**		**1.23333**		**10%**	**400**	**444.44**			**7 months**	**little change in body weight**	**body weight significantly decreased at 400 and 4,000 ppm PAHs at 3 months, but at 7 months change in growth was not significant; mixture with paraffin wax in food- aromatic mixture contained only some percentage of PAHs, not including benzo(a)pyrene**
PAHs (mixture)	400		Patton and Dieter 1980	mortality	Mallard	Petroleum hydrocarbon mixture including PAHs)	food	0.1218	W	2		1.218		10%	4000	4444.4			7 months	mortality	mixture with paraffin wax in food- aromatic mixture contained only some percentage of PAHs, not including benzo(a)pyrene; no statistics
Aldrin		**0.04**	**DeWitt 1956**	**mortality**	**Quail**	**aldrin**	**food**	**0.0048**	**D**	**1**	**3**	**0.09**	**B**	**10%**			**0.5**	**0.56**	**5 mos**	**97.5% mortality in 127 days**	**no stats provided**
Aldrin		0.7	DeWitt 1956	mortality	Pheasant	aldrin	food	0.0464	D	1	3	1.135	B	10%			5	5.6	16-20 wks	100% mortality in 46 days	no stats provided
Aldrin	10	20	Hall et al. 1971	growth	pen-reared pheasant chicks	aldrin	gelatin capsule					0.007							7 wks (+ 9 wks untreated)	growth	dose calculation is highly uncertain, birds given one dose per week; growth significantly reduced in birds fed 1.0 mg in capsule; no effect on birds fed 0.5 mg

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Table 3. Summary of Updated Bird Dietary TRV Review for Round 2 ERA

Analyte	NOAEL (mg/kg bw/d)	LOAEL (mg/kg bw/d)	Source	Endpoint	Test Species	Chemical Form	Exposure Mode	FI (kg dw or L/day)	Wet or Dry?	FI Default?	Nagy bird guild	Body Weight (kg)	BW Default?	% Moisture	NEC wet (ppm)	NEC dry (ppm)	LEC wet (ppm)	LEC dry (ppm)	Exposure Duration	Effect Endpoint	Notes
Aldrin		92	Hall et al. 1971	mortality	pen-reared pheasant chicks	aldrin	gelatin capsule					0.007							7 wks (+ 9 wks untreated)	growth	dose calculation is highly uncertain; birds given a dose 3 X per wk; 8/22 birds died after single 1.5 mg dose (no stats); mortality at other doses not clearly reported
Methoxychlor	**34.6**	**346**	**Gee et al. 2004**	**reproduction**	**Zebra finch (chicks at 5-11 days old)**	**technical methoxychlor**	**oral gavage**												**1 week**	**number of eggs hatched, broken/missing eggs**	**borderline effect on egg fertility at 346 mg/kg bw/d**
Methoxychlor	**34.6**	**346**	**Millam et al. 2002**	**mortality**	**Zebra finch chicks**	**methoxychlor**	**oral gavage**												**1 week**	**mortality**	**significant mortality at LOAEL - 28.2% (13 to 46 died); significant increase in growth also observed at mortality LOAEL**
Methoxychlor	831		Hill et al. 1975b; Heath et al. 1972	mortality	Bobwhites (juvenile), Japanese quail, Ringed-necked pheasant, mallard (young)	technical methoxychlor	food	0.0150	D	1	1	0.1	A	10%	5000	5555.6			5 days	mortality	0% mortality

NC = TRV not calculated in database because more preferable studies were available for TRV selection (see notes)
FI = food ingestion rate
NEC = No effect concentration in exposure medium
LEC = Low effect concentration in exposure medium
W = wet weight basis
D = dry weight basis

Default ingestion rates:
1 - Nagy 2001
2 - Heinz et al. 1987
3 - NRC 1984
4 - NRC 1994
5 - EPA 1993
6 - Edens and Garlich 1983
7 - Kushlan 1978

Nagy bird group allometric equation
1- all birds: FI (kg/d dw) = [0.638*((bw(g))^0.685)]/1000
2- Passerines: FI = [0.630*((bw(g))^0.683)]/1000
3- Galliformes: FI = [0.088*((bw(g))^0.891)]/1000
4- Omnivorous birds: FI = [0.670*((bw(g))^0.627)]/1000
5- Carnivorous birds: FI = [0.849*((bw(g))^0.663)]/1000
6- Eurasian Kestrel: FI =(22.1/211)*bw(kg)

Default body weight:
A - NRC 1994
B - Dunning 1993
C - NRC 1984
D - Peakall et al. 1975
E - Sample et al. 1996
F - EPA 1993
G - Edens and Garlich 1983
H - Arizona Game and Fish 2002
I - Pattee 1984

(text in red and cells highlighted in orange and with red text indicate changes in the TRV from Appendix B of the RI/FS (submitted to EPA on September 9, 2005))
Shaded rows indicate TRV studies added to the TRV review process since the preparation of Appendix B of the PRE (submitted to EPA on September 9, 2005)
Bolded rows indicate selected Round 2 TRVs

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Table 4. Summary of Updated Mammal Dietary TRV Review for Round 2 ERA

Analyte	NOAEL (mg/kg bw/d)	LOAEL (mg/kg bw/d)	Source	Endpoint	Test Species	Exposure Mode	FI (kg dw or L/day)	Wet or Dry?	FI Default?	Body Weight (kg)	BW Default?	% Moisture	NEC wet (ppm)	NEC dry (ppm)	LEC wet (ppm)	LEC dry (ppm)	Chemical Form	Exposure Duration	Effect Endpoint	Notes
Selenium		0.08	**Halverson et al. 1966**	**growth**	**Sprague-Dawley rat**	**food**	**0.00215**	**W**		**0.129**					4.8		**$NaSeO_3$ or seleniferous wheat**	**6 wks**	**body weight**	**statistically significant effects at LOAEL**
Selenium	0.055		**Halverson et al. 1966**	**growth**	**Sprague-Dawley rat**	**food**	**0.00238**	**W**		**0.139**			3.2				**$NaSeO_3$ or seleniferous wheat**	**6 wks**	**body weight**	**statistically evaluated**
Selenium		0.14	Halverson et al. 1966	mortality	Sprague-Dawley rat	food	0.00186	W		0.1255					8.0		$NaSeO_3$ or seleniferous wheat	6 wks		no statistics for mortality
Selenium	0.03		Halverson et al. 1966	mortality	Sprague-Dawley rat	food	0.00215	W		0.129			8				$NaSeO_3$ or seleniferous wheat	6 wks		no statistics for mortality
Selenium		0.16	Behne et al. 1992	growth	Wistar rat	food	0.027	W	1	0.34					2		L-selenomethionine	110 days	body weight	
Selenium	0.16		Behne et al. 1992	growth	Wistar rat	food	0.027	W	1	0.34			2				selenite	110 days	body weight	
Selenium	0.16	0.27	Rosenfeld and Beath 1954	reproduction	Wistar rat	drink water	0.038485	W	2	0.35	A		1.5		2.5		$KSeO_3$	1 yr	pup growth, number weaned	statistics not presented
Selenium	0.17	0.28	Jia et al. 2005	growth	Sprague-Dawley rat	food											Na_2SeO_3, Nano-Se, or organic selenium	13 wks	body weight	
Selenium	0.36		Julius et al. 1983	growth	Hamster	food	0.00655	W		0.092			5.1				selenomethionine	21d	body weight	no statistics
Selenium	0.42		Schroeder and Mitchener 1971	mortality	Charles River mouse	drink water	0.004217	W	2	0.03	A		3				SeO_4	multigenerational	survival	
Selenium		0.43	Schroeder and Mitchener 1971	reproduction	Charles River mouse	drink water	0.004217	W	2	0.03	A				3.06		SeO_4	multigenerational	offspring mortality, number of runts, breeding failure	
Selenium		0.43	Schroeder and Mitchener 1972	mortality, growth	Swiss mouse	drink water	0.004217	W	2	0.03	A				3.06		Na_2SeO_4	lifetime	survival, body weight	
Selenium		0.76	Julius et al. 1983	growth	Hamster	food	0.0068	W		0.091					10.1		selenomethionine	21d	body weight	no statistics
Selenium		0.87	Ferm et al. 1990	growth	Hamster	injection (iv)											sodium selenite (NaSeO3)	single injection during gestation	maternal body weight	statistically significant effects at LOAEL
Selenium	1.82		Ferm et al. 1990	reproduction	Hamster	injection (iv)											sodium selenite (NaSeO3)	single injection during gestation	number of abnormal litters	statistically significant effects at LOAEL
Selenium	2.21	2.68	Ferm et al. 1990	growth, reproduction	Hamster	injection (iv)											sodium selenate (Na2SeO4)	single injection during gestation	maternal body weight, number of abnormal litters	statistically significant effects at LOAEL
Selenium		3.4	Julius et al. 1983	growth	Hamster	food	0.0062	W		0.07375					40.25		Na_2SeO_3	21d	body weight	no statistics
Selenium		5.8	Julius et al. 1983	mortality	Hamster	food	0.0045	W		0.062					80.25		Na_2SeO_3	21 d	female mortality	no statistics
Selenium		7.11	Ferm et al. 1990	reproduction	Hamster	gavage											sodium selenate (Na2SeO4)	4 days of gestation	fetal body length	statistically significant effects at LOAEL
Selenium	7.11	7.90	Ferm et al. 1990	reproduction	Hamster	gavage											sodium selenite (NaSeO3)	4 days of gestation	number of abnormal litters, fetal body length	statistically significant effects at LOAEL
Selenium	7.90	8.69	Ferm et al. 1990	growth	Hamster	gavage											sodium selenate (Na2SeO4)	4 days of gestation	maternal body weight	statistically significant effects at LOAEL
Zinc	191		Llobet et al. 1988	growth	Rat	drink water											$Zn(CH_3COO)_2 \cdot 2H_2O$	3 mos	body weight	Zn aversion suspected; drinking water exposure
Zinc	80	400	Sutton and Nelson 1937	reproduction	Rat	food	0.028	W	1	0.35	A	10%	1000	1111.111	5000	5556	zinc carbonate	gestation	pregnancy, pup survival and body weight	no statistics
Zinc	160	320	**Schlicker and Cox 1968**	**reproduction**	**Sprague-Dawley rat**	**food**	**0.028**	**W**	**1**	0.35		**10%**	**2000**	**2222.222**	**4000**	**4444**	**zinc oxide**	**gestation**	**fetal growth, number of resorptions**	**statistically significant effects at LOAEL**
Zinc	149	433	Straube et al. 1980	growth	Ferret	food	0.17	W		0.6		76%	527	2195.833	1527	6362.5	zinc oxide	2wks-6 mos		no statistics; exposure period variable
Zinc	400	799	Sutton and Nelson 1937	growth	Rat	food	0.028	W	1	0.35	A	10%	5000	5555.556	10000	11111	zinc carbonate	gestation	body weight	no statistics
Zinc	433	858	Straube et al. 1980	mortality	Ferret	food	0.17	W		0.6		76%	1527	6362.5	3027	12612.5	zinc oxide	2wks-6 mos		no statistics; exposure period variable; mortality endpoint measured incorrectly
DBT	**3.8**	**7.6**	**Harazono and Ema 2003**	**growth**	**Wistar rat**	**gavage**											**dibutyltin dichloride**	**3 days during pregnancy**	**maternal body weight**	

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Table 4. Summary of Updated Mammal Dietary TRV Review for Round 2 ERA

Analyte	NOAEL (mg/kg bw/d)	LOAEL (mg/kg bw/d)	Source	Endpoint	Test Species	Exposure Mode	FI (kg dw or L/day)	Wet or Dry?	FI Default?	Body Weight (kg)	BW Default?	% Moisture	NEC wet (ppm)	NEC dry (ppm)	LEC wet (ppm)	LEC dry (ppm)	Chemical Form	Exposure Duration	Effect Endpoint	Notes
DBT		**7.6**	**Ema et al. 2003**	**reproduction, growth**	**Wistar rat**	**gavage**											**dibutyltin dichloride**	**3 days during pregnancy**	**reduced no. implantations, no. of females pregnant, increased rate of postimplantation loss, reducedmaternal body weight**	
TBT	**0.4**	**2.0**	**Omura et al. 2001**	**reproduction**	**Wistar rat**	**food**							**5**		**25**		**tributyltin chloride**	**Multi-generational**	**pup body weight**	
TBT		**2.0**	**Makita et al. 2003; Makite et al. 2004**	**reproduction**	**Wistar rat**	**food**									**25**		**tributyltin chloride**	**reproduction period**	**pup body weight**	
TBT	0.21	2.1	Wester et al. 1990	growth, mortality	Wistar rat	food	0.016	W		0.39	A		5		50		bis (tri-n-butyltin) oxide	106 wks	male body weight gain, adult mortality	
TBT	2.0	10	Ogata et al. 2001	reproduction	Wistar rat	food							25		125		tributyltin chloride	Multi-generational	pup body weight, percentage of live pups	
TBT		5	Baroncelli et al. 1995	reproduction	Swiss mouse	gavage											bis (tri-n-butyltin) oxide	days 6-15 of gestation	number of pups/ implantations	statistically significant effect; most sensitive endpoint measured
TBT	5	10	Crofton et al. 1989	growth, reproduction	Long-Evans rat	gavage (oral)											tributyltin oxide	day 6- 20 of gestation	maternal body weight, litter size, pup body weight, increased age of sexual maturity	statistically evaluated effects
TBT		10	Ema et al. 1995a	growth, mortality, reproduction	Wistar rat	gavage (stomach)											di-butyltin chloride	day 7 and 8 of gestation	fetal body weight, embryo lethality, fetus abnormalities, adult mortality, maternal body weight	most sensitive endpoints measured
TBT	23.4	35	Davis et al. 1987	reproduction	mouse	gavage (stomach)											tributyltin oxide	day 6-15 of gestation	number of living fetuses per litter, resorptions, fetal body weight	no statistics- at LOAEL 59% resorptions, 5% fetal survival and fetal body weight = 0.91g; at NOAEL 10.1% resorptions, 11.6% fetal survival and fetal body weight = 1.04 g; at control 9.1% resorptions, 11.5% fetal survival and fetal body weight = 1.13g
TBT	20	40	Baroncelli et al. 1990	growth, reproduction	Swiss albino mouse	gavage											bis (tri-n-butyltin) oxide	days 6-15 of gestation	number of live fetuses, fetal body weight, fetal malformations, maternal growth	significant decrease in placental weight at 5 mg/kg/d (not reported as LOAEL)
TBT		40	Ema et al. 1995a	growth, reproduction	Wistar rat	gavage (stomach)											tributyltin chloride	day 7 and 8 of gestation	post-implantation loss, maternal body weight change during pregnancy	most sensitive endpoints measured
TBT		47.8	Ema et al. 1995c	growth, reproduction	Wistar rat	gavage											TBTCl	days 13-15 of gestation	fetal malformations, maternal growth	statistically significant effects at unbounded LOAEL
TBT	5.8	58 - 146	Davis et al. 1987	growth	mouse	gavage (stomach)											tributyltin oxide	day 6-15 of gestation	body weight	no statistics- 10-40% reduction in growth at LOAEL range, "no effect" on growth at NOAEL, growth data not reported- qualitatively presented
TBT		1000	Ema et al. 1995a	growth	Wistar rat	gavage (stomach)											butyltin chloride	day 7 and 8 of gestation	maternal body weight change during pregnancy	most sensitive endpoint measured
Aldrin	**0.83**	**4.1**	**Reuber 1980**	**mortality**	**Osborne-Mendel rat**	**food**	**0.026**	**W**	**1**	**0.32**		**10%**	**10**	**11**	**50**	**55.6**	**aldrin**	**2 yrs**		**no statistics**
Dieldrin	0.050		Walker et al. 1969b	growth	Beagle dog	oral capsule											dieldrin	2 yrs	body weight	
Dieldrin		0.19	Treon and Cleveland 1955	reproduction	Carworth rat	food	0.030	W	1	0.40		10%			2.5	2.8	dieldrin	3 generations	offspring mortality	no statistics; data not presented
Dieldrin	0.5		Walker et al. 1969b	mortality, growth	CFE rat	food	0.017	W		0.33			9.66	9.66			dieldrin	2 yrs		no dose-response for mortality: 1ppm~0.0475 and 0.05825 mg/kg/d (male and female)
Dieldrin	0.92		Good and Ware 1969	mortality	CFS Swiss mouse	food	0.006	W	1	0.03	A	10%	5	5.555556			tech dieldrin	120 d		
Dieldrin		**0.92**	**Good and Ware 1969**	**reproduction**	**CFS Swiss mouse**	**food**	**0.006**	**W**	**1**	**0.03**	**A**	**10%**			**5**	**5.6**	**tech dieldrin**	**120 d**	**litter size**	
Dieldrin		**0.92**	**Virgo and Bellward 1977**	**reproduction**	**mouse**	**food**	**0.006**	**W**	**1**	**0.03**	**A**				**5**		**dieldrin**	**4 wks prior to mating through gestation**	**18% pup survival**	**Statistically significant effects (pup survival at control = 35%)**
Dieldrin	0.9	1.8	Virgo and Bellward 1977	behavior	mouse	food	0.006	W	1	0.03	A		5		10		dieldrin	4 wks prior to mating through gestation	maternal behavior (tendency/ onset of nursing)	Statistically significant effects; endpoint not direct measure of reproductive success
Dieldrin		1.8	Thorpe and Walker 1973	mortality	CFI mouse	food	0.006	W	1	0.03	A	10%			10	11.1	dieldrin	2 yrs		mortality results not clear; liver tumors significant at 10 ppm

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Table 4. Summary of Updated Mammal Dietary TRV Review for Round 2 ERA

Analyte	NOAEL (mg/kg bw/d)	LOAEL (mg/kg bw/d)	Source	Endpoint	Test Species	Exposure Mode	FI (kg dw or L/day)	Wet or Dry?	FI Default?	Body Weight (kg)	BW Default?	% Moisture	NEC wet (ppm)	NEC dry (ppm)	LEC wet (ppm)	LEC dry (ppm)	Chemical Form	Exposure Duration	Effect Endpoint	Notes
Dieldrin		1.84	Blidstein and Forsyth 1979	behavior	White-footed mouse	food	0.006	W	1	0.03	A				10		dieldrin	approximately 3 months	altered anti-predator response to hawk	only one dose given, effect in number of mouse with no response increased from the number with no response in the control, and the number responding by freezing was significantly less than the number responding by freezing in control
Dieldrin	1.3	2.0	Treon and Cleveland 1955	mortality	Dogs	food							10		25		dieldrin	~9-15 mos		no statistics, no control; small sample size
Dieldrin	2.2		Treon and Cleveland 1955	growth	Carworth rat	food	0.023	W	1	0.26		10%	25	28			dieldrin	6 mos		
Dieldrin	0.8	4.1	Tremuth et al. 1964; Haubar 1980	mortality	Osborne-Mendel rat	food	0.026	W	1	0.32		10%	10	11	50	55.6	dieldrin	2 yrs		no statistics
Dieldrin	6.6		Treon and Cleveland 1955	mortality	Carworth rat	food	0.023	W	1	0.26		10%	75	83			dieldrin	6 mos		
Dieldrin	12		Fitzhugh et al. 1964	growth	Osborne-Mendel rat	food	0.026	W	1	0.32		10%	150	167			dieldrin	2 yrs	body weight	
Dieldrin		15	Walsh and Fink 1972	mortality	CF-1 mouse	injection											technical grade dieldrin	single injection, effects observed for 24 hrs	study measured time to death (LT50); LC50 = 10.5 mg/kg; mortality observed at all doses administered	no control
Dieldrin		26	Treon and Cleveland 1955	mortality	Carworth rat	food	0.023	W	1	0.26		10%			300	333.3	dieldrin	6 mos		
Dieldrin	1.9		Treon and Cleveland 1955	mortality	Carworth rat	food	0.030	W	1	0.40		10%	25	28			dieldrin	2 yrs		
Methoxychlor		5.0	Chapin et al. 1997	growth	rat	gavage											methoxychlor	gestation day 17-PND 7	dam body weight	dose units assumed body weight-normalized
Methoxychlor		7.1	Gray et al. 1989	growth	Long Evans rat	gavage											methoxychlor	weaning to 85 days	male body weight	dose frequency =2 times per week
Methoxychlor	7.1	14	Gray et al. 1989	reproduction	Long Evans rat	gavage											methoxychlor	weaning to 85 days	female fecundity	dose frequency =2 times per week
Methoxychlor		33	Amstislavsky et al. 1999	reproduction	mouse	injection (sc)											methoxychlor	gestation day 5-7	pregnancy rate, number of offspring, sexual arousal and motivation	only one dose used, pregnancy rate = 40% (80-100% in control); decreased number of offspring (statistically significant)
Methoxychlor	25	50	Gray et al. 1988	growth	rat	gavage											methoxychlor	~ 11 mos (weaning-lactation)	male body weight at necropsy	statistics not described, control data not presented; not enough detail; exposure route unclear
Methoxychlor		50	Chapin et al. 1997	reproduction	rat	gavage											methoxychlor	parents thru gestation and pnd 7, pups pnd 7-42	offspring fertility	dose units assumed body weight-normalized
Methoxychlor		**56**	**You et al. 2002**	**growth, reproduction**	**Sprague-Dawley rat**	**food**	**0.016**	**W**		**0.23**					**800**		**methoxychlor**	**gestation day 0-pnd 22 (adult), pnd 28-100 young**	**body weight; offspring growth rate, litter size, sex development of offspring**	**statistically significant effects at LOAEL**
Methoxychlor	14 - 71	68 - 357	Swartz and Eroschenko 1998	reproduction	mouse	injection (ip)											methoxychlor	14 days, prior to pregnancy	number of resorptions/ litter	LOAEL and NOAEL are range of mg/kg bw dose reported in study; injections were daily for 14 days
Methoxychlor		80	Harris et al. 1974	reproduction	Sprague-Dawley rat	food	0.028	W	1	0.35	A	10%			1000	1111.1111	tech methoxychlor	parents treated, weanlings treated 8 wks thru mating	% mated, % littered	
Methoxychlor	17	86	Masutomi et al. 2003	reproduction, growth	Sprague-Dawley rat	food	0.028	W	1	0.35	A		216	240	1080	1200	methoxychlor	gestation day 15, postnatal day 10	maternal body weight, offspring body weight, delayed onset of puberty	offspring and maternal body weight was signicantly lower during experiment but was recovered by end of experiment
Methoxychlor	50	100	Gray et al. 1988	reproduction	rat	gavage											methoxychlor	~ 11 mos (weaning-lactation)	litter size, fertility	statistics not described, control data not presented; not enough detail; other reproductive endpoints (less relevant); exposure route unclear
Methoxychlor		100	Bal 1984	reproduction	Wistar rat	gavage											gamma HCH	14 days	infertile females, inhibition of spermatogenesis and folliculogenesis	impairment of reproductive success
Methoxychlor		168	Harris et al. 1974	reproduction	Sprague-Dawley rat	food	0.016	W		0.24					2500		tech methoxychlor	pre-mating thru weaning	% mated, litter production and size	
Methoxychlor	100	200	Cummings and Gray 1989	reproduction	Holtzman rat	gavage											methoxychlor	gestation day 1-8	number of implantations	
Methoxychlor	167	250	Swartz and Corkern 1992	reproduction	CD-1 mouse	gavage				0.03	A		5		7.5		tech methoxychlor	gestation day 6-15	fetal mortality	
Methoxychlor	500		Cummings and Gray 1989	growth	Holtzman rat	gavage											methoxychlor	gestation day 1-8	body weight gain	
Phenol	**60**	**120**	**Argus Research Laboratories 1997 (as cited in IRIS 2006)**	**growth**	**Rat**	**Gavage**												**Gestational days 6 - 15**	**maternal body weight**	**dose 3X per day; Original citation not obtained**
Phenol	**60**	**120**	**Charles River Laboratories 1988 and NTP 1983a (as cited in IRIS 2006)**	**Reproduction**	**Rat**	**Gavage**													**decreased fetal body weight**	**Original citation not obtained**
Phenol	140	280	NTP 1983b (as cited in IRIS 2006)	mortality, growth	Mouse	gavage												Gestational days 6 - 15	mortality and reduced body weight	Original citation not obtained

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Table 4. Summary of Updated Mammal Dietary TRV Review for Round 2 ERA

Analyte	NOAEL (mg/kg bw/d)	LOAEL (mg/kg bw/d)	Source	Endpoint	Test Species	Exposure Mode	FI (kg dw or L/day)	Wet or Dry?	FI Default?	Body Weight (kg)	BW Default?	% Moisture	NEC wet (ppm)	NEC dry (ppm)	LEC wet (ppm)	LEC dry (ppm)	Chemical Form	Exposure Duration	Effect Endpoint	Notes

NC = TRV not calculated in database because more preferable studies were available for TRV selection
NEC = No effect concentration in vehicle
LEC = Low effect concentration in vehicle
W = wet weight basis
D = dry weight basis
FI = food ingestion rate
DWI = drinking water ingestion rate

Text in red and cells highlighted in orange and with red text indicate changes in the TRV from Appendix B of the PRE (submitted to EPA on September 9, 2005).
Shaded rows indicate TRV studies added to the TRV review process since the preparation of Appendix B of the PRE (submitted to EPA on September 9, 2005).

Default intake rates:
1 - EPA 1993, laboratory mammals: FI (kg/d ww) = 0.058*(bw (kg))^0.8611;
2 - Calder and Braun 1983, all mammals = DWI (L/d) = 0.0691*(bw (kg))^0.9;
3 - Beavers and Aulerich 1981

Default body weight:
A - EPA 1993
B - Beavers and Aulerich 1981

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APPENDIX G: ECOLOGICAL RISK ASSESSMENT

Attachment G6
Screening Assessment
for Wildlife

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1.0 INITIAL SCREENING PROCESS

In the initial screening process for wildlife receptors, all lines of evidence associated with the two measures of exposure and effects were evaluated (see Section 5.0 of Appendix G). These two measures were: dietary dose estimates compared to dietary dose toxicity reference values (TRVs) and estimated bird egg tissue concentrations compared to bird egg tissue TRVs. These two measures of exposure and effects included an initial screening process, which is presented in the following sections.

2.0 DIETARY DOSE ASSESSMENT

The primary line of evidence for evaluating risks to all wildlife receptors was the dietary dose assessment. In the dietary dose assessment line of evidence, daily dietary doses were quantitatively estimated for each chemical and wildlife receptor. Dietary dose estimates included ingestion of biota (i.e., prey) and incidental ingestion of sediment.

Dietary doses for the identification of chemicals of potential concern (COPCs) were estimated using the following equation:

$$IR_{diet} = \frac{\left[(FIR \times C_{prey}) + (SIR \times C_{sed})\right]}{BW} \times SUF \qquad \textbf{Equation 1}$$

Where:

IR_{diet}	=	estimated bird or mammal exposure dose or intake rate (mg/kg bw/day)
FIR	=	food ingestion rate (kg ww food/day)
C_{prey}	=	tissue concentration in prey items (mg/kg ww)
SIR	=	sediment ingestion rate (kg dw sediment/day)
C_{sed}	=	concentration in surface sediment (mg/kg dw)
BW	=	body weight (kg)
SUF	=	site use factor (unitless); fraction of time that a receptor spends foraging in the Study Area relative to their entire home range; an SUF of 1.0 was assumed for all receptors in the screening assessment.

2.1 DIETARY SCREEN AND COPC IDENTIFICATION

To identify COPCs for wildlife, the maximum exposure dose (IR_{diet}) of each chemical of interest (COI) was compared to a no-observed-adverse-effect-level (NOAEL) TRV for that chemical. If the maximum exposure dose (for each receptor) was greater than the NOAEL TRV, the chemical was identified as a COPC.

The identification of COPCs through a conservative screen in which maximum concentrations in relevant media are used is consistent with EPA guidance (EPA 1997a, b) and an important step for narrowing the list of chemicals to only those that could potentially pose a risk to ecological receptors.

2.1.1 Selection of COIs

COIs for wildlife were defined as all chemicals detected in surface sediment and fish or invertebrate tissue from the ERA dataset (Table 2-1), excluding crustal elements (i.e., aluminum and manganese; see Section 2.0 of Appendix G).

Total PCBs were defined as a COI for both birds and mammals. PCB risks to birds and mammals were evaluated in two ways: as exposure to total PCBs and as exposure to dioxin-like PCB congeners. Dioxin-like PCB congeners show structural and toxic similarities to dioxins and furans. The relative toxicity of each individual dioxin-like PCB congener and 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (2,3,7,8-TCDD) is quantified using toxic equivalency factors (TEFs). These factors and the concentrations of each dioxin-like PCB congener are then summed to calculate a toxic equivalent (TEQ), as follows:

$$TEQ = \sum_{i=1}^{n} C_i TEF_i \qquad \textbf{Equation 2}$$

Where:

TEF_i = TEF for an individual dioxin, furan, or PCB congener
C_i = concentration of an individual fish prey tissue (mg/kg ww)

Using the assumption that the combined effect of these congeners is additive, the total TCDD-like toxicity is estimated by summing the TEF concentration products for individual PCB congeners. This TEQ approach was also used to evaluate risks associated with dioxins and furans. A PCB TEQ and dioxin TEQ (including both dioxins and furans) were calculated to estimate exposure doses for birds and mammals. Attachment G1 presents the bird and mammal TEFs that were used to derive TEQs. The TEFs used in this ERA were developed by the World Health Organization (WHO) in 1998 (Van den Berg et al. 1998) from a database that contains all available mammal, bird, and fish studies involving the relative toxicity of dioxin-like compounds. WHO has recently updated mammal TEFs (Van den Berg et al. 2006). Mammal exposure doses of TEQs (both PCB TEQs and dioxin TEQs) based on these updated TEFs were calculated and are presented in the uncertainty section presented in Section 5.0 of Appendix G.

The TEQ approach accounts for the toxicity of coplanar PCB congeners that have dioxin-like modes of toxic action. PCB congeners from other structural classes may elicit other toxic responses with different toxic mechanisms. Therefore, the TEQ approach is not used as a surrogate for the total PCB approach, which captures all PCB toxicity mechanisms or modes of action, because it contains multiple classes of congeners including coplanar PCB congeners. Uncertainties in the TEQ approach for evaluating risks associated with dioxin-like PCB congeners and risks with dioxins and furans include limitations in the underlying data used to derive TEFs, the relevancy of the endpoints in the studies, and possible interspecies variability. These uncertainties were evaluated and are presented in the uncertainty section presented in Section 4.0 of Appendix G.

Additional organic chemicals that were detected in surface sediment but not detected in fish or invertebrate tissue (Table 2-2) were not evaluated as COIs because biota exposure is the primary route of chemical exposure for wildlife receptors. Six of the

chemicals in Table 2-2 have been identified as important bioaccumulative compounds (EPA 2000): Mirex, toxaphene, 1,2,4-trichlorobenzene, 1,2-dichlorobenzene, 1,3-dichlorobenzene, and 1,4-dichlorobenzene. However, none of these chemicals were detected in biological samples from the Study Area. In addition, all six of these chemicals were detected in < 4% of surface sediment samples. These two factors indicate that these chemicals are not present in significant quantities in the Study Area, and are not bioaccumulating in biological tissues.

Table 2-3 presents additional chemicals that were detected in sediment but not analyzed in tissue. None of these additional chemicals have been identified as important bioaccumulative compounds (EPA 2000) and none were evaluated as COIs for wildlife receptors.

2.1.2 Identification of COPCs

COPCs were identified for each wildlife receptor by comparing estimated maximum exposure doses (see Equation 1) for all COIs to NOAEL TRVs. If the maximum exposure dose for a receptor was greater than the NOAEL TRV, the chemical and receptor were identified as a COPC/receptor pair for this Round 2 ERA.

Bird and mammal NOAEL TRVs were developed for all wildlife COIs when literature was available. NOAEL TRVs were based on an extensive search on the available toxicological literature for the development of bird and mammal dietary TRVs and were selected to represent conservative screening level toxicological thresholds. These NOAEL TRVs were presented in the Ecological PRE (Windward 2005). EPA's comments to the Ecological PRE state that "overall, [avian and mammalian dietary] TRVs selected were the most conservative of those acceptable studies and are reflective of EPA's comments for deriving adequately conservative dietary TRVs" (EPA 2006b). EPA provided further direction (EPA 2006a, d) on specific wildlife dietary dose TRVs that should be changed for this Round 2 ERA (e.g., the use of EPA's Ecological Soil Screening Levels [Eco SSLs] for metals, when data were available), and these changes were incorporated to the NOAEL TRVs. Several additional COIs have been identified since the compilation of TRVs for the Ecological PRE; and additional toxicological studies for these new COIs, as well as some additional literature for existing COIs, were incorporated into the process for TRV selection. The TRV technical memorandum addendum (Attachment G5) presents the results of the entire literature search process, a detailed discussion of how TRVs were derived, and recommended TRVs for this Round 2 report.

Tables 2-4 and 2-5 present those COIs for bird and mammals, respectively. COIs where no toxicological data (i.e., no NOAEL TRV) were identified are discussed in the uncertainty section presented in Section 5.0 of Appendix G. No bird NOAEL TRVs (or TRVs based on a surrogate chemicals) were identified for ten COIs: antimony, silver, 2-methylnaphthalene, hexachloroethane, 2-methylphenol, 4-methylphenol, phenol, benzyl alcohol, dibenzofuran, and n-nitrosodiphenylamine. No mammal NOAEL TRVs (or TRVs based on a surrogate chemicals) were identified

for six COIs: antimony, silver, 2-methylphenol, 4-methylphenol, benzyl alcohol, dibenzofuran, and n-nitrosodiphenylamine.

Selected NOAEL TRVs are based on the NOAEL TRVs recommended in the TRV Technical Memorandum (Appendix B of the Ecological PRE (Windward 2005)), based on EPA's comments to the TRVs (EPA 2006a, b, d), and based on the TRV technical memorandum addendum (Attachment G5). For COIs where no chronic NOAEL TRVs were recommended, (i.e., where no NOAEL was selected, or the selected NOAEL was not based on a chronic study), uncertainty factors (UFs) were used to derive chronic NOAELs, EPA and its partners recommended using UFs to develop conservative TRVs in a data screening evaluation (EPA 2005). UFs for the Round 2 ERA were based on EPA Region 10 guidance and used to extrapolate a TRV where a chronic NOAEL was not selected (Table 2-6). UFs, also called safety factors, are often used to provide conservative screening levels and/or risk estimates when more specific toxicological information is lacking. They are intended to account for the extrapolation of toxicity data to a desired effect level, exposure duration, and species, and to account for uncertainty in extrapolating to conditions with no data (Duke and Taggart 2000). Tables 2-7 and 2-8 present the selected dietary dose NOAEL TRVs for birds and mammals, respectively.

To identify COPCs for birds and mammals, NOAEL TRVs were compared to the estimated maximum dietary dose (IR_{diet}) for each of the six wildlife receptors. The maximum dietary exposure dose was calculated as milligrams of each COPC ingested per kilogram of body weight per day (mg/kg bw/day) using Equation 5-1. The maximum daily ingested doses for sandpiper, hooded merganser, bald eagle, osprey, mink, and river otter were calculated using the maximum concentrations in any of the appropriate prey species in the ERA dataset. Table 2-9 presents the appropriate prey species identified for each receptor. The maximum surface sediment concentration (representing C_{sed}) and the maximum tissue concentration (representing C_{prey}) in any of the prey species identified for each receptor (Table 2-9) were used to estimate maximum exposure dose (IR_{diet}). The exposure parameters (body weight, food ingestion rate, and sediment ingestion rate) and dietary prey items used to calculate the maximum dietary exposure doses for each wildlife receptor are presented in Table 2-10. In accordance with EPA comments (EPA 2006b), dietary doses for all wildlife receptors were based on female exposure parameters. An SUF of 1.0 was assumed for all wildlife receptors for the identification of COPCs.

Estimates of the maximum daily ingested doses of 11 metals (arsenic, cadmium, chromium, copper, lead, mercury, selenium, thallium, zinc, tributyltin, and dibutyltin), and 12 organics (dioxin TEQ, PCB TEQ, total PCBs, total PAHs, benzo(a)pyrene, aldrin, sum DDD, sum DDE, sum DDT, total DDTs, bis [2-ethylhexyl] phthalate [BEHP], and dibutylphthalate) were greater than their respective NOAEL TRVs or Eco SSLs for spotted sandpiper (Table 2-11). These chemicals were retained as COPCs for spotted sandpiper.

Estimates of the maximum daily ingested doses of 3 metals (copper, lead, and mercury), and 8 organics (dioxin TEQ, PCB TEQ, total PCBs, benzo(a)pyrene, sum DDE, sum DDT, total DDTs, and BEHP) were greater than their respective NOAEL TRVs or Eco SSLs for hooded merganser (Table 2-12). These chemicals were retained as COPCs for hooded merganser.

Estimates of the maximum daily ingested doses of 2 metals (lead and mercury), and 6 organics (dioxin TEQ, PCB TEQ, total PCBs, sum DDE, sum DDT, and BEHP) were greater than their respective NOAEL TRVs or Eco SSLs for bald eagle (Table 2-13). These chemicals were retained as COPCs for bald eagle.

Estimates of the maximum daily ingested doses of 2 metals (lead and mercury), and 7 organics (dioxin TEQ, PCB TEQ, total PCBs, benzo(a)pyrene, sum DDE, sum DDT, and BEHP) were greater than their respective NOAEL TRVs or Eco SSLs for osprey (Table 2-14). These chemicals were retained as COPCs for osprey.

Estimates of the maximum daily ingested doses of 5 metals (antimony, copper, lead, mercury, and selenium), and 5 organics (dioxin TEQ, PCB TEQ, total PCBs, total PAHs, and total DDTs) were greater than their respective NOAEL TRVs or Eco SSLs for mink (Table 2-15). These chemicals were retained as COPCs for mink.

Estimates of the maximum daily ingested doses of 5 organics (dioxin TEQ, PCB TEQ, total PCBs, total PAHs, and total DDTs) were greater than their respective NOAEL TRVs for river otter (Table 2-16). These chemicals were retained as COPCs for river otter.

A summary of the COPCs that were evaluated for each wildlife receptor are presented in Table 2-17.

3.0 BIRD EGG ASSESSMENT

As an additional line of evidence for evaluating risks to piscivorous birds (i.e., osprey and bald eagle), risks to specific COIs were evaluated by estimating concentrations in egg tissues and comparing estimated concentrations to bird egg TRVs from the literature. Egg tissue concentrations were estimated using prey tissue concentrations and prey tissue-to-egg tissue biomagnification factors (BMFs). This line of evidence was evaluated to be protective of developing embryos and a sensitive life-stage.

Chemical concentrations in bird eggs are not available from the Study Area; however, bird egg tissue residue concentrations were estimated using the following equation:

$$EPC_{egg} = BMF \times C_{tiss}$$ **Equation 3-1**

Where:

C_{egg} = estimated exposure concentration in bird egg tissue (mg/kg ww)
BMF = biomagnification factor
C_{tiss} = tissue concentration in fish prey item (mg/kg ww)

BMFs were specific to each chemical, and were selected from region-specific literature.

3.1 BIRD EGG SCREEN AND COPC IDENTIFICATION

To identify COPCs for using the bird egg approach, the maximum exposure egg concentration (EPC_{egg}) of each COI was compared to a NOAEL TRV for that chemical. If the maximum exposure dose (for both osprey and bald eagle) was greater than the NOAEL, the chemical was identified as a COPC.

The identification of COPCs through a conservative screen where maximum concentrations in relevant media are used is consistent with EPA Guidance (cite Guidance), and an important step for narrowing the list of chemicals to only those that may potentially pose a risk to ecological receptors.

3.1.1 Selection of COIs

COIs for the bird egg approach were limited to specific bioaccumulative chemicals agreed upon by LWG and EPA in the preparation of the Ecological PRE (Windward 2005). The following chemicals were evaluated as COIs for piscivorous birds using the bird egg approach:

- Dioxins and furans (evaluated as dioxin TEQ for birds)
- Dioxin-like PCBs (evaluated as PCB TEQ for birds)
- PCBs (evaluated as total PCBs)

- DDT and metabolites (evaluated as DDE)
- Mercury

3.1.2 Identification of COPCs

COPCs were identified by comparing maximum estimated bird egg tissue concentrations for all COIs to a NOAEL TRV. If the maximum bird egg tissue concentration was greater than the NOAEL TRV, the chemical was identified as a COPC for bird eggs for this Round 2 ERA.

Bird egg NOAEL TRVs were developed for all bird egg COIs, where literature was available. NOAEL TRVs were based on an extensive search on the available toxicological literature for the development of bird egg TRVs and were selected to represent conservative screening level toxicological thresholds. These NOAEL TRVs were presented in the Ecological PRE. EPA's comments to the Ecological PRE state that "(bird egg) TRVs derived for non-domestic species more specific to the Willamette River should be used for the next phase of the risk assessment…" (EPA 2006b). EPA further clarified that the selected screening level TRVs for bird eggs presented in the Ecological PRE may be overly conservative for bird receptors in the Study Area (EPA 2006a). For the purposes of the COI screen and COPC identification, the NOAEL TRVs recommended by LWG in the Ecological PRE were used. The results of the TRV literature search and a detailed discussion of how TRVs were derived are presented in the TRV Technical Memorandum (Appendix B of the PRE). NOAEL TRVs were developed for all bird egg COIs and are presented in Table 3-1. Selected NOAEL TRVs are based on the NOAEL TRVs recommended in the TRV Technical Memorandum (Appendix B of the Ecological PRE).

For the purposes of the risk characterization of COPCs, bird egg TRVs based on non-domestic species specific to the Willamette River (in accordance with EPA comments) were selected and used. LWG supports EPA's comments that TRVs based on more representative species should be used for COPC risk characterization in this Round 2 ERA and in the BERA.

To identify COPCs in bird eggs, NOAEL TRVs were compared to the estimated maximum egg tissue concentration. The maximum bird egg tissue concentration for piscivorous birds was calculated using the maximum fish concentration in the ERA dataset to represent C_{prey}. The BMFs used to calculate the maximum bird egg tissue concentration are presented in Table 3-2.

Estimates of the all five bird egg COIs (dioxin TEQ, PCB TEQ, total PCBs, DDE, and mercury) were greater than their respective NOAEL TRVs. These chemicals were all retained as COPCs for osprey and bald eagles.

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TABLES

Table 2-1. Wildlife COIs

Chemicals of Interest	
Metals	
Antimony	Mercury
Arsenic	Nickel
Cadmium	Selenium
Chromium	Silver
Copper	Thallium
Lead	Zinc
Butyltins	
Butyltin ion	Tetrabutyltin[a]
Dibutyltin ion[a]	Tributyltin ion
PAHs	
2-Methylnaphthalene	Chrysene
Acenaphthene	Dibenzo(a,h)anthracene
Acenaphthylene	Fluoranthene
Anthracene	Fluorene
Benzo(a)anthracene	Indeno(1,2,3-cd)pyrene
Benzo(a)pyrene	Naphthalene
Benzo(b)fluoranthene	Phenanthrene
Benzo(b+j)fluoranthene[b]	Pyrene
Benzo(g,h,i)perylene	Total PAHs
Benzo(k)fluoranthene	
Phthalates	
Bis(2-ethylhexyl) phthalate	Diethyl phthalate[a]
Butylbenzyl phthalate[c]	Dimethyl phthalate[a]
Dibutyl phthalate	Di-n-octyl phthalate[c]
SVOCs	
Benzyl alcohol	Hexachlorobutadiene
Dibenzofuran	Hexachloroethane[c]
Hexachlorobenzene	n-Nitrosodiphenylamine[a]
Phenols	
2-Methylphenol[a]	Pentachlorophenol
4-Methylphenol	Phenol
PCBs	
Total PCBs	
Dioxins and Furans	
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin	1,2,3,6,7,8-Hexachlorodibenzofuran
2,3,7,8-Tetrachlorodibenzofuran	1,2,3,7,8,9-Hexachlorodibenzofuran
1,2,3,7,8-Pentachlorodibenzo-*p*-dioxin	2,3,4,6,7,8-Hexachlorodibenzofuran
2,3,4,7,8-Pentachlorodibenzofuran	1,2,3,4,6,7,8-Heptachlorodibenzo-*p*-dioxin
1,2,3,7,8-Pentachlorodibenzofuran	1,2,3,4,6,7,8-Heptachlorodibenzofuran

Table 2-1. Wildlife COIs

Chemicals of Interest	
1,2,3,4,7,8-Hexachlorodibenzo-*p*-dioxin	1,2,3,4,7,8,9-Heptachlorodibenzofuran
1,2,3,6,7,8-Hexachlorodibenzo-*p*-dioxin	Octachlorodibenzo-*p*-dioxin
1,2,3,7,8,9-Hexachlorodibenzo-*p*-dioxin	Octachlorodibenzofuran
1,2,3,4,7,8-Hexachlorodibenzofuran	
Pesticides	
Aldrin	Endosulfan sulfate
cis-Chlordane	Endrin
trans-Chlordane	Endrin aldehyde
Total chlordane	Endrin ketone
2,4′-DDD	Heptachlor
2,4′-DDE	Heptachlor epoxide
2,4′-DDT	alpha-Hexachlorocyclohexane
4,4′-DDD	beta-Hexachlorocyclohexane
4,4′-DDE	gamma-Hexachlorocyclohexane
4,4′-DDT	delta-Hexachlorocyclohexane
Total DDTs	Methoxychlor
Dieldrin	cis-Nonachlor
alpha-Endosulfan	trans-Nonachlor
beta-Endosulfan	Oxychlordane
Total Endosulfan	

[a] These COIs were detected in invertebrate tissue and surface sediment only.
[b] Benzo(B+J)fluoranthene was detected in fish tissue only.
[c] These COIs were detected in fish tissue and surface sediment only.
COI – chemical of interest
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound

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Table 2-2. Chemicals Detected in Surface Sediment[a] but Not Detected in Fish or Invertebrate Tissue

Chemicals	
Phenols	
2,3,4,5-Tetrachlorophenol	2,4-Dichlorophenol
2,3,5,6-Tetrachlorophenol	2,4-Dimethylphenol
2,4,5-Trichlorophenol	2-Chlorophenol
2,4,6-Trichlorophenol	4-Chloro-3-methylphenol
SVOCs	
1,2,4-Trichlorobenzene	Aniline
1,2-Dichlorobenzene	Benzoic acid
1,3-Dichlorobenzene	Bis(2-chloroethyl) ether
1,4-Dichlorobenzene	Carbazole
4-Chloroaniline	3-Nitroaniline
4-Nitroaniline	
Pesticides	
Mirex	Toxaphene

[a] The surface sediment Study Area dataset for ERA is defined in Section 2.0.
SVOC – semivolatile organic compound

Table 2-3. Chemicals Detected in Surface Sediment[a] but Not Analyzed in Fish or Invertebrate Tissue

Chemicals	
Metals	
Barium	Potassium[b]
Beryllium	Sodium[b]
Calcium[b]	Tin[b]
Cobalt[b]	Titanium[b]
Iron[b]	Vanadium[b]
Magnesium[b]	
PAHs	
1,6,7-Trimethylnaphthalene[b]	C2-Fluorene[b]
1-Methylnaphthalene[b]	C2-Naphthalene[b]
1-Methylphenanthrene[b]	C2-Phenanthrene/anthracene[b]
2,6-Dimethylnaphthalene[b]	C3-Chrysene[b]
Benzo(b+k)fluoranthene[b]	C3-Dibenzothiophene[b]
Benzo(e)pyrene[b]	C3-Fluoranthene/pyrene[b]
Benzo(j+k)fluoranthene[b]	C3-Fluorene[b]
C1-Chrysene[b]	C3-Naphthalene[b]
C1-Dibenzothiophene[b]	C3-Phenanthrene/anthracene[b]
C1-Fluoranthene/pyrene[b]	C4-Chrysene[b]
C1-Fluorene[b]	C4-Dibenzothiophene[b]
C1-Naphthalene[b]	C4-Naphthalene[b]
C1-Phenanthrene/anthracene[b]	C4-Phenanthrene/anthracene[b]
C2-Chrysene[b]	Dibenzothiophene[b]
C2-Dibenzothiophene[b]	Perylene[b]
C2-Fluoranthene/pyrene[b]	
SVOCs	
Diphenyl[b]	Bis(2-chloroisopropyl) ether
Herbicides	
2,4,5-T[c]	Dichloroprop
2,4-D	MCPA
2,4-DB	MCPP
2,4,5-T[c]	Silvex
VOCs	
1,1,1,2-Tetrachloroethane	Isopropylbenzene
1,1,2,2-Tetrachloroethane	Methyl isobutyl ketone
1,1-Dichloroethane	Methyl N-butyl ketone
1,2,3-Trichloropropane	Methyl tert-butyl ether
1,2-Dichloroethane	Methylene chloride
cis-1,2-Dichloroethene	Methylethyl ketone
trans-1,2-Dichloroethene	Styrene
Acetone	Tetrachloroethene

Table 2-3. Chemicals Detected in Surface Sediment[a] but Not Analyzed in Fish or Invertebrate Tissue

Chemicals	
Benzene	Toluene
Carbon disulfide	Trichloroethene
Chlorobenzene	Vinyl chloride
Chloroethane	m,p-Xylene
Chloroform	o-Xylene
Dichlorodifluoromethane	Total xylenes
Ethylbenzene	
TPH	
Diesel-range hydrocarbons	Motor oil[b]
Gasoline-range hydrocarbons	Residual-range hydrocarbons
Lube oil[b]	Total petroleum hydrocarbons

[a] The surface sediment Study Area dataset for ERA is defined in Section 2.0.
[b] Detected in non-LWG collected surface sediment only (not analyzed in LWG collected surface sediment).
[c] Detected in non-LWG collected surface sediment only (not detected in LWG collected surface sediment).
MCPA – (4-chloro-2-methylphenoxy)acetic acid
MCPP – 2-(4-chloro-2-methylphenoxy)-propanoic acid
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound
TPH – total petroleum hydrocarbons
VOC – volatile organic compound

Table 2-4. Results of TRV Search for COIs for Birds

COIs with TRVs	
Metals	
Arsenic	Mercury
Cadmium	Nickel
Chromium	Selenium
Copper	Thallium
Lead	Zinc
Butyltins	
Butyltin ion[a]	Tetrabutyltin[a]
Dibutyltin ion[a]	Tributyltin ion
PAHs	
Benzo(a)pyrene	Total PAHs[b]
Phthalates	
Bis(2-ethylhexyl) phthalate	Diethyl phthalate[a]
Butylbenzyl phthalate[a]	Dimethyl phthalate[a]
Dibutyl phthalate[a]	Di-n-octyl phthalate[a]
SVOCs	
Hexachlorobenzene	Hexachlorobutadiene
Phenols	
Pentachlorophenol	
PCBs	
Total PCBs	PCB TEQ
Dioxins and Furans	
Dioxin TEQ	
Pesticides	
Aldrin	Endrin
Total chlordane[c]	Endrin aldehyde[a]
Sum DDD (2,4′-DDD + 4,4′-DDD)	Endrin ketone[a]
Sum DDE (2,4′-DDE + 4,4′-DDE)	Heptachlor
Sum DDT (2,4′-DDT + 4,4′-DDT)	Heptachlor epoxide[a]
Total DDTs[d]	alpha-Hexachlorocyclohexane[a]
Dieldrin	beta-Hexachlorocyclohexane[a]
alpha-Endosulfan	gamma-Hexachlorocyclohexane
beta-Endosulfan	delta-Hexachlorocyclohexane[a]
Total endosulfan[c]	Methoxychlor
Endosulfan sulfate	

Table 2-4. Results of TRV Search for COIs for Birds

COIs without TRVs	
Antimony	4-Methylphenol
Silver	Phenol
2-Methylnaphthalene	Benzyl alcohol
Hexachloroethane	Dibenzofuran
2-Methylphenol	n-Nitrosodiphenylamine

[a] A surrogate chemical was used to represent a TRV for this COI.

[b] Total PAHs is a sum including the following COIs: acenaphthene, acenaphthylene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(b+j)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenzo(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3-cd)pyrene, phenanthrene, and pyrene.

[c] Total chlordane is a sum including the following COIs: cis-Chlordane, trans-Chlordane, Oxychlordane, cis-Nonachlor, and trans-Nonachlor.

[d] Total DDT is a sum including the following COIs: 2,4′-DDD, 2,4′-DDE, 2,4′-DDT, 4,4′-DDD, 4,4′-DDE, and 4,4′-DDT.

[c] Total endosulfan is a sum including the following COIs: alpha-endosulfan, gamma-endosulfan, and endosulfan sulfate.

COI – chemical of interest
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TEQ – toxic equivalent
TRV – toxicity reference value

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Table 2-5. Results of TRV Search for COIs for Mammals

COIs with TRVs	
Metals	
Antimony	Mercury
Arsenic	Nickel
Cadmium	Selenium
Chromium	Thallium
Copper	Zinc
Lead	
Butyltins	
Butyltin ion[a]	Tetrabutyltin[a]
Dibutyltin ion	Tributyltin ion
PAHs	
2-Methylnaphthalene	Naphthalene
Benzo(a)pyrene	Total PAHs[a,b]
Phthalates	
Bis(2-ethylhexyl) phthalate	Diethyl phthalate
Butylbenzyl phthalate	Dimethyl phthalate[a]
Dibutyl phthalate	Di-n-octyl phthalate
SVOCs	
Hexachlorobenzene	Hexachloroethane
Hexachlorobutadiene	
Phenols	
Pentachlorophenol	Phenol
PCBs	
Total PCBs	PCB TEQ
Dioxins and Furans	
Dioxin TEQ	
Pesticides	
Aldrin	Endrin ketone[a]
Total chlordane[c]	Heptachlor
Total DDTs[d]	Heptachlor epoxide[a]
Dieldrin	alpha-Hexachlorocyclohexane
alpha-Endosulfan	beta-Hexachlorocyclohexane
Total endosulfan[c]	gamma-Hexachlorocyclohexane
beta-Endosulfan	delta-Hexachlorocyclohexane
Endosulfan sulfate	Methoxychlor
Endrin	trans-Nonachlor
Endrin aldehyde[a]	
COIs without TRVs	
Silver	Benzyl alcohol
2-Methylphenol	Dibenzofuran
4-Methylphenol	n-Nitrosodiphenylamine

[a] A surrogate chemical was used to represent a TRV for this COI.
[b] Total PAHs is a sum including the following COIs: acenaphthene, acenaphthylene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(b+j)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenzo(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene, and pyrene.
[c] Total chlordane is a sum including the following COIs: cis-chlordane, trans-chlordane, oxychlordane, cis-nonachlor, and trans-nonachlor.
[d] Total DDT is a sum of including following COIs: 2,4′-DDD, 2,4′-DDE, 2,4′-DDT, 4,4′-DDD, 4,4′-DDE, and 4,4′-DDT. Sum DDD, sum DDE, and sum DDT were not evaluated for wildlife receptors because insufficient toxicity data were available for these DDT summations.
[c] Total endosulfan is a sum including the following COIs: alpha-endosulfan, gamma-endosulfan, and endosulfan sulfate.
COI – chemical of interest
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TEQ – toxic equivalent
TRV – toxicity reference value

Table 2-6. Uncertainty Factors Used to Derive Chronic NOAEL TRVs for Birds and Mammals

Source	cLOAEL: cNOAEL	sLOAEL: cNOAEL	aLOAEL (non-lethal): cNOAEL	aLOAEL (lethal; LC50): cNOAEL
EPA Region 10 Guidance (EPA 1997b)	5	10[a]	10[a]	50

[a] For acute or subchronic LOAEL to chronic NOAEL
aLOAEL – acute LOAEL
cLOAEL – chronic LOAEL
cNOAEL – chronic NOAEL
sLOAEL – subchronic LOAEL
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
TRV – toxicity reference value

Table 2-7. Bird Dietary NOAEL TRVs Used for Screening to Determine COPCs

COI	Species	NOAEL TRV (mg/kg bw/day)	Endpoint	Source
Metals				
Arsenic	na	2.24	na	EPA Ecological SSL[a]
Cadmium	na	1.47	na	EPA Ecological SSL[a]
Chromium	na	2.66	na	EPA Ecological SSL[a]
Copper	na	4.05	na	EPA Ecological SSL[a]
Lead	na	1.63	na	EPA Ecological SSL[a]
Mercury	mallard	0.010[b]	reproduction, behavior	Heinz (1975; 1979)
Nickel	mallard	77	survival, growth	Cain and Pafford (1981)
Selenium	mallard	0.42	reproduction	Heinz et al. (1989)
Thallium	pheasant	0.48[c]	survival	Hudson et al. (1984)
Zinc	chicken	82	growth	Roberson and Schaible (1960)
Butyltins				
Monobutyltin	Japanese quail	1.4[d]	reproduction	Schlatterer et al. (1993); Coenen et al. (1992)
Dibutyltin	Japanese quail	1.4[d]	reproduction	Schlatterer et al. (1993); Coenen et al. (1992)
Tetrabutyltin	Japanese quail	1.4[d]	reproduction	Schlatterer et al. (1993); Coenen et al. (1992)
Tributyltin	Japanese quail	1.4	reproduction	Schlatterer et al. (1993); Coenen et al. (1992)
PAHs				
Benzo(a)pyrene	pigeon	0.28[b]	reproduction	Hough et al. (1993)
Total PAHs	mallard	40	growth	Patton and Dieter (1980)
Phthalates				
BEHP	ringed turtle dove	1.45	reproduction	Peakall (1974)
Butyl benzyl phthalate	ringed turtle dove	1.45[c]	reproduction	Peakall (1974)
Dibutyl phthalate	ringed turtle dove	1.45[c]	reproduction	Peakall (1974)
Diethyl phthalate	ringed turtle dove	1.45[c]	reproduction	Peakall (1974)
Dimethyl phthalate	ringed turtle dove	1.45[c]	reproduction	Peakall (1974)
Di-n-octyl phthalate	ringed turtle dove	1.45[c]	reproduction	Peakall (1974)

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Table 2-7. Bird Dietary NOAEL TRVs Used for Screening to Determine COPCs

COI	Species	NOAEL TRV (mg/kg bw/day)	Endpoint	Source
SVOCs				
Hexachlorobenzene	Japanese quail	0.24[b]	reproduction	Schwetz et al. (1974)
Hexachlorobutadiene	Japanese quail	1.7	growth, reproduction	Schwetz et al. (1974)
Phenols				
Pentachlorophenol	chicken	22	growth	Prescott et al. (1982)
PCBs				
PCB TEQ	pheasant	0.000014	growth, survival, reproduction	Nosek et al. (1992)
Total PCBs	chicken	0.29	reproduction	Britton and Huston (1973)
Dioxins and Furans				
Dioxin TEQ	pheasant	0.000014	growth, survival, reproduction	Nosek et al. (1992)
Pesticides				
Aldrin	quail	0.008[b, f]	survival	DeWitt (1956)
Total chlordane	bobwhite quail	0.6	growth, survival	Ludke (1976)
Sum DDE	barn owl	0.064[b,l]	reproduction	Mendenhall et al. (1983)
Sum DDD	mallard	0.180[b]	reproduction	Heath et al. (1969)
Sum DDT	quail	0.030[b]	reproduction	Stickel and Rhodes (1970)
Total DDTs	mallard	0.18	reproduction	Davison and Sell (1974)
Dieldrin	barn owl	0.066	survival, reproduction	Mendenhall et al. (1983)
alpha-Endosulfan	gray partridge	10[g]	reproduction	Abiola (1992)
beta-Endosulfan	gray partridge	10[g]	reproduction	Abiola (1992)
Total endosulfan	gray partridge	10	reproduction	Abiola (1992)
Endosulfan sulfate	gray partridge	10[g]	reproduction	Abiola (1992)
Endrin	quail	0.012[b]	reproduction	DeWitt (1956)
Endrin aldehyde	quail	0.012[i]	reproduction	DeWitt (1956)
Endrin ketone	quail	0.012[i]	reproduction	DeWitt (1956)
Heptachlor	bobwhite (juvenile)	0.1[c]	survival	Hill et al. (1975); Heath et al. (1972)
Heptachlor epoxide	bobwhite (juvenile)	0.1[j]	survival	Hill et al. (1975); Heath et al. (1972)
alpha-HCH	mallard	1.6[h]	reproduction	Chakravarty and Lahiri (1986); Chakravarty et al.

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Table 2-7. Bird Dietary NOAEL TRVs Used for Screening to Determine COPCs

COI	Species	NOAEL TRV (mg/kg bw/day)	Endpoint	Source
				(1986)
beta-HCH	mallard	1.6[h]	reproduction	Chakravarty and Lahiri (1986); Chakravarty et al. (1986)
gamma-HCH	mallard	1.6	reproduction	Chakravarty and Lahiri (1986); Chakravarty et al. (1986)
delta-HCH	mallard	1.6[h]	reproduction	Chakravarty and Lahiri (1986); Chakravarty et al. (1986)
Methoxychlor	zebra finch (chicks)	34.6[i]	reproduction; survival	Gee et al. (2004); Millam et al. (2002)

[a] In response to EPA's comments (EPA 2006a), ecological soil screening levels (Eco SSLs) were used for screening COIs and identifying COPCs.
[b] Chronic NOAEL TRV was extrapolated from the chronic LOAEL TRV (see Appendix B Ecological PRE (Windward 2005)) using a UF of 5.
[c] Chronic NOAEL TRV was extrapolated from the acute lethal LD50 TRV using a UF of 50.
[d] NOAEL TRV for TBT was used as a surrogate.
[e] NOAEL TRV for BEHP was used as a surrogate.
[f] NOAEL TRVs were developed for this Round 2 ERA; details on the toxicity review of these new COIs are presented in the TRV Addendum in Attachment G5.
[g] NOAEL TRV for total endosulfan (based on technical endosulfan) was used as a surrogate.
[h] NOAEL TRV for gamma-HCH was used as a surrogate.
[i] NOAEL TRV for endrin was used as a surrogate.
[j] NOAEL TRV for heptachlor was used as a surrogate.
[k] NOAEL TRV reflects the additional toxicological literature reviewed since the submittal of the Ecological PRE (EPA 2006d) (see Attachment G5 for details).
[l] Chronic NOAEL TRV was extrapolated from the chronic LOAEL TRV (see Appendix B Ecological PRE (Windward 2005)) using a UF based on EPA's comments to the Ecological PRE (EPA 2006d).

bw – body weight
BEHP – bis(2-ethylhexyl) phthalate
COI – chemical of interest
COPC – chemical of potential interest
EPA – US Environmental Protection Agency
HCH – hexachlorocyclohexane
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyls
SVOC – semivolatile organic compound
TEQ – toxic equivalent
TRV – toxicity reference value

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Table 2-8. Mammal Dietary NOAEL TRVs Used for Screening to Determine COPCs

COI	Species	NOAEL TRV (mg/kg bw/day)	Endpoint	Source
Metals				
Antimony	na	0.059	na	EPA Ecological SSL[a]
Arsenic	na	1.04	na	EPA Ecological SSL[a]
Cadmium	na	0.77	na	EPA Ecological SSL[a]
Chromium	na	5.66	na	EPA Ecological SSL[a]
Copper	na	5.82	na	EPA Ecological SSL[a]
Lead	na	4.7	na	EPA Ecological SSL[a]
Mercury	mink	0.02	reproduction	Dansereau et al. (1999)
Nickel	rat	8.4	growth	Ambrose et al. (1976)
Selenium	rat	0.055	growth	Halverson et al. (1966)
Thallium	rat	0.74	growth	Formigli et al. (1986)
Zinc	rat	160	reproduction	Schlicker and Cox (1968)
Butyltins				
Monobutyltin	rat	0.4[b]	reproduction	Omura et al. (2001)
Dibutyltin	rat	3.8[c]	growth	Harazono and Ema (2003)
Tetrabutyltin	rat	0.4[b]	reproduction	Omura et al. (2001)
Tributyltin	rat	0.4[d]	reproduction	Omura et al. (2001)
PAHs				
2-Methylnaphthalene	mouse	54[c]	growth	Murata et al. (1997)
Benzo(a)pyrene	mouse	2.0[f,g,]	reproduction	MacKenzie and Angevine (1981)
Naphthalene	mouse	133	survival, growth	Shopp et al. (1984)
Total PAHs	mouse	1.0[h]	reproduction	MacKenzie and Angevine (1981)
Phthalates				
BEHP	mouse	44	reproduction	Tyl et al. (1988)
Butyl benzyl phthalate	rat	831	growth	Agarwal et al. (1985)
Dibutyl phthalate	rat	16[f]	reproduction	Wine et al. (1997)
Diethyl phthalate	mouse	1,860	growth, reproduction	Lamb et al. (1987)
Di-n-octyl phthalate	mouse	7,500	survival, growth, reproduction	Heindel et al. (1989)
Dimethyl phthalate	mouse	44[i]	reproduction	Tyl et al. (1988)
SVOCs				
Hexachlorobenzene	mink, ferret	0.026[f]	reproduction	Bleavins et al. (1984)
Hexachlorobutadiene	rat	2.0	survival, growth, reproduction	Kociba et al. (1977a; 1977b); Schwetz et al. (1977)
Hexachloroethane	rat	100	growth, reproduction	Weeks et al. (1979)

Table 2-8. Mammal Dietary NOAEL TRVs Used for Screening to Determine COPCs

COI	Species	NOAEL TRV (mg/kg bw/day)	Endpoint	Source
Phenols				
Phenol	rat	60[c]	growth, reproduction	Argus Research Laboratories 1997 and Charles River Laboratories 1988 and NTP 1983a (all as cited in IRIS [EPA 2006c])
Pentachlorophenol	rat	4.0	reproduction	Welsh et al. (1987)
PCBs				
Total PCBs	mink	0.0074[f]	reproduction	Restum et al. (1998)
PCB TEQ (mammals)	mink	0.00000044[f]	reproduction	Tillitt et al. (1996)
Dioxins and Furans				
Dioxin TEQ (mammals)	mink	0.00000044[f]	reproduction	Tillitt et al. (1996)
Pesticides				
Aldrin	rat	0.83[c]	survival	Reuber (1980)
Total chlordane	mouse	0.18	growth	Khasawinah and Grutsch (1989)
Total DDT	rat, mouse	0.26[g]	reproduction	Duby et al. (1971), Ware and Good (1967)
alpha-Endosulfan	mouse	0.84[j]	survival, growth	Hack et al. (1995)
beta-Endosulfan	mouse	0.84[j]	survival, growth	Hack et al. (1995)
Total endosulfan	mouse	0.84	survival, growth	Hack et al. (1995)
Dieldrin	mouse	0. 18[a]	reproduction	Good and Ware (1969); Virgo and Bellward (1977)
Endosulfan sulfate	mouse	0.84[j]	survival, growth	Hack et al. (1995)
Endrin	mouse	0.18[a]	survival, reproduction	Good and Ware (1969)
Endrin aldehyde	mouse	0. 18[l]	survival, reproduction	Good and Ware (1969)
Endrin ketone	mouse	0. 18[l]	survival, reproduction	Good and Ware (1969)
Heptachlor	mink	1.0	survival, growth	Crum et al. (1993)
Heptachlor epoxide	mink	1.0[m]	survival, growth	Crum et al. (1993)
alpha-HCH	rat	6.1[k]	survival, growth, reproduction	Palmer et al. (1978)
beta-HCH	rat	5.7	growth, survival	Van Velsen et al. (1986)
gamma-HCH	rat	6.1	survival, growth, reproduction	Palmer et al. (1978)
delta-HCH	rat	6.1[k]	survival, growth, reproduction	Palmer et al. (1978)
Methoxychlor	rat	5.6	growth, reproduction	You et al. (2002)
trans-Nonachlor	rat	2.5	growth, survival	Bondy et al. (2000)

[a] Per EPA's comments (EPA 2006a), ecological soil screening levels (Eco SSLs) were used for screening COIs and identifying COPCs.
[b] NOAEL TRV for TBT was used as a surrogate.
[c] NOAEL TRVs were developed for this Round 2 ERA and details on the toxicity review of these new COIs are presented in presented in the TRV Addendum in Attachment G5.
[d] NOAEL TRV reflects the additional toxicological literature reviewed since the submittal of the Ecological PRE (see Attachment G5 for details).
[e] NOAEL TRV reflects EPA's comments that no need of extrapolating from a subchronic NOAEL was necessary.
[f] Chronic NOAEL TRV was extrapolated from the chronic LOAEL TRV (see Appendix B Ecological PRE (Windward 2005)) using a UF of 5.
[g] Chronic NOAEL TRV was extrapolated from the chronic LOAEL TRV using a UF based on EPA's comments to the Ecological PRE (EPA 2006d).
[h] NOAEL TRV for benzo(a)pyrene was used as a surrogate.
[i] NOAEL TRV for BEHP was used as a surrogate.
[j] NOAEL TRV for total endosulfan (based on technical endosulfan) was used as a surrogate.
[k] NOAEL TRV for gamma-HCH was used as a surrogate.
[l] NOAEL TRV for endrin was used as a surrogate.
[m] NOAEL TRV for heptachlor was used as a surrogate.

BEHP – bis(2-ethylhexyl) phthalate
bw – body weight
COI – chemical of interest
COPC – chemical of potential concern
EPA – US Environmental Protection Agency
HCH – hexachlorocyclohexane
NOAEL – no-observed-adverse-effects level
PAH – polycyclic aromatic hydrocarbons
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
TRV – toxicity reference value

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Table 2-9. Prey Species Identified for Each Wildlife Receptor for Screening COIs

Receptor Species	Prey Item
Hooded merganser	clam; crayfish; sculpin; peamouth[a]
Spotted sandpiper	clam, worm[b]
Bald eagle	largescale sucker; carp; peamouth; northern pikeminnow[c]
Osprey	largescale sucker; northern pikeminnow;[c] carp; smallmouth bass; black crappie; brown bullhead
Mink	crayfish; largescale sucker; carp; sculpin; smallmouth bass; juvenile chinook salmon; peamouth;[a] northern pikeminnow;[c] brown bullhead; black crappie
River otter	crayfish; clam; carp; largescale sucker; sculpin; smallmouth bass; black crappie

[a] Sculpin concentrations of dioxin TEQ, PCB TEQ, PAHs, phenol, phthalates, and other SVOCs were used as a surrogate to represent these COPC concentrations for peamouth.
[b] Crayfish concentrations of thallium were used as a surrogate to represent thallium concentrations for laboratory worms.
[c] Smallmouth bass concentrations of dioxin TEQ, PCB TEQ, PAHs, phenol, phthalates, and other SVOCs were used as a surrogate to represent these COPC concentrations for northern pikeminnow.
COI – chemical of interest

Table 2-10. Exposure Parameters Used for Wildlife Dietary Risk Calculations

Receptor	BW (kg)	FIR (kg ww/ day)	SI (%)[a]	SIR (kg dw/ day)[b]
Spotted sandpiper	0.047	0.0548	18	0.00148
Hooded merganser	0.54	0.200	2	0.00110
Bald eagle	4.5	0.540	2	0.00281
Osprey	1.88	0.395	2	0.00205
Mink	0.974	0.156	9.4	0.00381
River otter	7.7	0.759	2	0.00465

[a] Percent of incidental sediment ingestion of the dry diet.
[b] SIR = FIR x SI.
BW – body weight
dw – dry weight
FIR – food ingestion rate
SI – sediment ingestion
SIR – sediment ingestion rate
ww – wet weight

Table 2-11. Results of COPC Screen for Spotted Sandpiper

COI	Exposure Concentration			Dietary Dose			COPC?
	Max C_{prey} (ww)	Max C_{sed} (dw)	Unit	Max IR_{diet}	NOAEL TRV/ Eco SSL	Unit (bw/day)	
Metals							
Arsenic	3.04	75.6	mg/kg	5.93	2.24	mg/kg	**yes**
Cadmium	0.254	46.2	mg/kg	1.75	1.47	mg/kg	**yes**
Chromium	1.05 T	774	mg/kg	25.6	2.66	mg/kg	**yes**
Copper	20.2	1,080	mg/kg	57.6	4.05	mg/kg	**yes**
Lead	0.847 T	1,950	mg/kg	62.4	1.63	mg/kg	**yes**
Mercury	0.016 J	4.84	mg/kg	0.171	0.01	mg/kg	**yes**
Nickel	0.535	594	mg/kg	19.3	77.4	mg/kg	no
Selenium	0.37	20	mg/kg	1.06	0.42	mg/kg	**yes**
Thallium	0.0079 J	27	mg/kg	0.859	0.48	mg/kg	**yes**
Zinc	54	2,850	mg/kg	153	82	mg/kg	**yes**
Butyltins							
Butyltin ion	3,600	740 J	µg/kg	4,220	1,400	µg/kg	**yes**
Dibutyltin ion	820	2,700	µg/kg	1,040	1,400	µg/kg	no
Tetrabutyltin	4.2	1,000	µg/kg	36.4	1,400	µg/kg	no
Tributyltin ion	1,700	47,000	µg/kg	3,460	1,400	µg/kg	**yes**
PAHs							
Benzo(a)pyrene	1,500	340,000	µg/kg	12,500	280	µg/kg	**yes**
Total PAHs	37,300	7,260,000	µg/kg	272,000	40,000	µg/kg	**yes**
Phthalates							
BEHP	340 UT	440,000 J	µg/kg	14,300	1,450	µg/kg	**yes**
Butylbenzyl phthalate	330 U	2,800	µg/kg	473	1,450	µg/kg	no
Dibutyl phthalate	1,300	3,790 J	µg/kg	1,640	1,450	µg/kg	**yes**
diethyl phthalate	160 U	2,150 UJ	µg/kg	254	1,450	µg/kg	no
Dimethyl phthalate	160 U	2,150 UJ	µg/kg	254	1,450	µg/kg	no
Di-n-octyl phthalate	640 U	15,400 JT	µg/kg	1,230	1,450	µg/kg	no
SVOCs							
Hexachlorobenzene	12 U	1,200 U	µg/kg	51.8	240	µg/kg	no
Hexachlorobutadiene	23 U	280 U	µg/kg	35.6	1,700	µg/kg	no
Phenols							
Pentachlorophenol	160 UT	8,410 J	µg/kg	451	22,000	µg/kg	no
PCBs							
PCB TEQ	732	5,610	pg/g	1,030	14	pg/g	**yes**
Total PCBs	4,310 J	30,800 J	µg/kg	5,995	290	µg/kg	**yes**
Dioxins and Furans							
Dioxin TEQ	932	35,300	pg/g	2,198	14	pg/g	**yes**
Pesticides							

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Table 2-11. Results of COPC Screen for Spotted Sandpiper

COI	Exposure Concentration			Dietary Dose			COPC?
	Max C_{prey} (ww)	Max C_{sed} (dw)	Unit	Max IR_{diet}	NOAEL TRV/ Eco SSL	Unit (bw/day)	
Aldrin	37	691 J	µg/kg	64.9	8.0	µg/kg	**yes**
cis-Chlordane	28.6	203 J	µg/kg	39.7	600	µg/kg	no
trans-Chlordane	30.5	445 NJ	µg/kg	49.6	600	µg/kg	no
Total chlordane	71.9	669 NJ	µg/kg	105	600	µg/kg	no
Sum DDD	1,260	3,040 NJ	µg/kg	1,560	180	µg/kg	**yes**
Sum DDE	188	2,530 NJ	µg/kg	300	64	µg/kg	**yes**
Sum DDT	114	12,500 NJ	µg/kg	527	30	µg/kg	**yes**
Total DDTs	1,490	16,200 NJ	µg/kg	2,250	180	µg/kg	**yes**
Dieldrin	26.7	356 J	µg/kg	42.3	66	µg/kg	no
alpha-Endosulfan	1 U	99 U	µg/kg	4.28	10,000	µg/kg	no
beta-Endosulfan	2.1 UT	235 NJ	µg/kg	9.85	10,000	µg/kg	no
Endosulfan sulfate	1	240 T	µg/kg	8.72	10,000	µg/kg	no
Total endosulfan	2.1 U	270	µg/kg	11.0	10,000	µg/kg	no
Endrin	2.4 UT	200 U	µg/kg	9.10	12	µg/kg	no
Endrin aldehyde	1 UJ	200 U	µg/kg	7.46	12	µg/kg	no
Endrin ketone	1 U	200 U	µg/kg	7.46	12	µg/kg	no
Heptachlor	1 U	99 U	µg/kg	4.28	100	µg/kg	no
Heptachlor epoxide	8 UT	99 U	µg/kg	12.4	100	µg/kg	no
alpha-HCH	1.06 U	99 U	µg/kg	4.35	1,600	µg/kg	no
beta-HCH	8.5 UT	99 U	µg/kg	13.0	1,600	µg/kg	no
delta-HCH	1 U	99 U	µg/kg	4.28	1,600	µg/kg	no
gamma-HCH	1 U	430	µg/kg	14.7	1,600	µg/kg	no
Methoxychlor	1.6 UJ	990 U	µg/kg	33.0	34,600	µg/kg	no
cis-Nonachlor	12 U	378 NJ	µg/kg	25.9	600	µg/kg	no
trans-Nonachlor	9.8	226 NJ	µg/kg	18.5	600	µg/kg	no
Oxychlordane	1.13 U	130 U	µg/kg	5.41	600	µg/kg	no

BEHP – bis(2-ethylhexyl) phthalate
bw – body weight
COI – chemical of interest
COPC – chemical of potential concern
dw – dry weight
HCH – hexachlorocyclohexane
IR – ingestion rate
J – estimated concentration
N – presumptive evidence of a compound
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
T – value calculated or selected from multiple results
TEQ – toxic equivalent
TRV – toxicity reference value
U – not detected at given concentration
ww – wet weight

Bold indentifies when maximum estimated dietary dose is greater than the NOAEL TRV or Eco SSL.

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Table 2-12. Results of COPC Screen for Hooded Merganser

COI	Exposure Concentrations			Dietary Dose			COPC?
	Max Cprey (ww)	Max Csed (dw)	Unit	Max IR_{diet}	NOAEL TRV/ Eco SSL	Unit (bw/day)	
Metals							
Arsenic	1.25	75.6	mg/kg	0.617	2.24	mg/kg	no
Cadmium	0.218	46.2	mg/kg	0.175	1.47	mg/kg	no
Chromium	1.05 T	774	mg/kg	1.97	2.66	mg/kg	no
Copper	17.6	1,080	mg/kg	8.72	4.05	mg/kg	**yes**
Lead	10.6	1,950	mg/kg	7.90	1.63	mg/kg	**yes**
Mercury	0.086	4.84	mg/kg	0.0417	0.01	mg/kg	**yes**
Nickel	0.83	594	mg/kg	1.52	77.4	mg/kg	no
Selenium	0.4	20	mg/kg	0.189	0.42	mg/kg	no
Thallium	0.012	27	mg/kg	0.0594	0.48	mg/kg	no
Zinc	54	2,850	mg/kg	25.8	82	mg/kg	no
Butyltins							
Butyltin ion	97	740 J	µg/kg	37.4	1,400	µg/kg	no
Dibutyltin ion	560	2,700	µg/kg	213	1,400	µg/kg	no
Tetrabutyltin	3.4 U	1,000	µg/kg	3.30	1,400	µg/kg	no
Tributyltin ion	530	47,000	µg/kg	292	1,400	µg/kg	no
PAHs							
Benzo(a)pyrene	490	340,000	µg/kg	874	280	µg/kg	**yes**
Total PAHs	4,980	7,260,000	µg/kg	16,600	40,000	µg/kg	no
Phthalates							
BEHP	28,000 JT	440,000 J	µg/kg	11,300	1450	µg/kg	**yes**
Butylbenzyl phthalate	370 U	2,800	µg/kg	143	1,450	µg/kg	no
Dibutyl phthalate	1,300	3,790 J	µg/kg	489	1,450	µg/kg	no
Diethyl phthalate	950 U	2,150 UJ	µg/kg	356	1,450	µg/kg	no
Dimethyl phthalate	190 U	2,150 UJ	µg/kg	74.8	1,450	µg/kg	no
Di-n-octyl phthalate	1,300 U	15,400 JT	µg/kg	513	1,450	µg/kg	no
SVOCs							
Hexachlorobenzene	33 UT	1,200 U	µg/kg	14.7	240	µg/kg	no
Hexachlorobutadiene	33 UT	280 U	µg/kg	12.8	1,700	µg/kg	no
Phenols							
Pentachlorophenol	190 UT	8,410 J	µg/kg	87.5	22,000	µg/kg	no
PCBs							
PCB TEQ	117	5,610	pg/g	54.8	14	pg/g	**yes**
Total PCBs	3,400	30,800 J	µg/kg	1,320	290	µg/kg	**yes**
Dioxins and Furans							
Dioxin TEQ	166	35,300	pg/g	133	14	pg/g	**yes**
Pesticides							

Table 2-12. Results of COPC Screen for Hooded Merganser

COI	Exposure Concentrations			Dietary Dose			COPC?
	Max Cprey (ww)	Max Csed (dw)	Unit	Max IR_{diet}	NOAEL TRV/ Eco SSL	Unit (bw/day)	
Aldrin	13 UT	691 J	µg/kg	6.22	8.0	µg/kg	no
cis-Chlordane	20 U	203 J	µg/kg	7.82	600	µg/kg	no
trans-Chlordane	9.6 U	445 NJ	µg/kg	4.46	600	µg/kg	no
Total chlordane	23.2 NJ	669 NJ	µg/kg	9.96	600	µg/kg	no
Sum DDD	376	3,040 NJ	µg/kg	145	180	µg/kg	no
Sum DDE	657	2,530 NJ	µg/kg	248	64	µg/kg	**yes**
Sum DDT	2,030	12,500 NJ	µg/kg	777	30	µg/kg	**yes**
Total DDTs	3,060	16,200 NJ	µg/kg	1,170	180	µg/kg	**yes**
Dieldrin	24 JT	356 J	µg/kg	9.61	66	µg/kg	no
alpha-Endosulfan	20 U	99 U	µg/kg	7.61	10,000	µg/kg	no
beta-Endosulfan	15 UT	235 NJ	µg/kg	6.03	10,000	µg/kg	no
Total Endosulfan	20 U	270	µg/kg	7.96	10,000	µg/kg	no
Endosulfan sulfate	12 UT	240 T	µg/kg	4.93	10,000	µg/kg	no
Endrin	31 UT	200 U	µg/kg	11.9	12	µg/kg	no
Endrin aldehyde	9.6 U	200 U	µg/kg	3.96	12	µg/kg	no
Endrin ketone	9.6 U	200 U	µg/kg	3.96	12	µg/kg	no
Heptachlor	13 UT	99 U	µg/kg	5.02	100	µg/kg	no
Heptachlor epoxide	9.6 U	99 U	µg/kg	3.76	100	µg/kg	no
alpha-HCH	9.6 U	99 U	µg/kg	3.76	1,600	µg/kg	no
beta-HCH	9.6 U	99 U	µg/kg	3.76	1,600	µg/kg	no
gamma-HCH	9.6 U	430	µg/kg	4.43	1,600	µg/kg	no
delta-HCH	9.6 U	99 U	µg/kg	3.76	1,600	µg/kg	no
Methoxychlor	9.6 U	990 U	µg/kg	5.57	34,600	µg/kg	no
cis-Nonachlor	12 U	378 NJ	µg/kg	5.21	600	µg/kg	no
trans-Nonachlor	11 UT	226 NJ	µg/kg	4.53	600	µg/kg	no
Oxychlordane	20 UT	130 U	µg/kg	7.67	600	µg/kg	no

BEHP – bis(2-ethylhexyl) phthalate
bw – body weight
COI – chemical of interest
COPC – chemical of potential interest
dw – dry weight
Eco SSL – ecological soil screening level
HCH – hexachlorocyclohexane
IR – ingestion rate
J – estimated concentration
N – presumptive evidence of a compound
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyls
SVOC – semivolitile organic compound
T – value calculated or selected from multiple results
TEQ – toxic equivalent
TRV – toxicity reference value
U – not detected at given concentration
ww – wet weight

Bold indentifies when the maximum estimated dietary dose is greater than the NOAEL TRV or Eco SSL.

Table 2-13. Results of COPC Screen for Bald Eagle

COI	Exposure Concentrations			Dietary Dose			COPC?
	Max C_{prey} (ww)	Max C_{sed} (dw)	Unit	Max IR_{diet}	NOAEL TRV/ Eco SSL	Unit (bw/day)	
Metals							
Arsenic	0.48	75.6	mg/kg	0.105	2.24	mg/kg	no
Cadmium	0.108	46.2	mg/kg	0.0418	1.47	mg/kg	no
Chromium	2.77 T	774	mg/kg	0.816	2.66	mg/kg	no
Copper	1.61	1,080	mg/kg	0.868	4.05	mg/kg	no
Lead	10.6	1,950	mg/kg	2.49	1.63	mg/kg	**yes**
Mercury	0.494	4.84	mg/kg	0.0623	0.01	mg/kg	**yes**
Nickel	1.37 J	594	mg/kg	0.535	77.4	mg/kg	no
Selenium	0.4	20	mg/kg	0.0605	0.42	mg/kg	no
Thallium	0.0093	27	mg/kg	0.0180	0.48	mg/kg	no
Zinc	112	2,850	mg/kg	15.2	82	mg/kg	no
Butyltins							
Butyltin ion	97[a]	740 J	µg/kg	12.1	1,400	µg/kg	no
Dibutyltin ion	560[a]	2,700	µg/kg	68.9	1,400	µg/kg	no
Tetrabutyltin	3.4[a] U	1,000	µg/kg	1.03	1,400	µg/kg	no
Tributyltin ion	530[a]	47,000	µg/kg	92.9	1,400	µg/kg	no
PAHs							
Benzo(a)pyrene	33 UT	340,000	µg/kg	216	280	µg/kg	no
Total PAHs	523	7,260,000	µg/kg	4,600	40,000	µg/kg	no
SVOCs							
Hexachlorobenzene	32 UT	1,200 U	µg/kg	4.59	240	µg/kg	no
Hexachlorobutadiene	9.8 UJ	280 U	µg/kg	1.35	1,700	µg/kg	no
Phenols							
Pentachlorophenol	170 UT	8,410 J	µg/kg	25.7	22,000	µg/kg	no
Phthalates							
BEHP	87,000 JT	440,000 J	µg/kg	10,700	1,450	µg/kg	**yes**
Butylbenzyl phthalate	660 U	2,800	µg/kg	80.9	1,450	µg/kg	no
Dibutyl phthalate	660 U	3,790 J	µg/kg	81.6	1,450	µg/kg	no
diethyl phthalate	1,600 U	2,150 UJ	µg/kg	193	1,450	µg/kg	no
Dimethyl phthalate	330 U	2,150 UJ	µg/kg	40.9	1,450	µg/kg	no
Di-n-octyl phthalate	2,100 T	15,400 JT	µg/kg	262	1,450	µg/kg	no
PCBs							
PCB TEQ	117	5,610	pg/g	17.5	14	pg/g	**yes**
Total PCBs	7,460 J	30,800 J	µg/kg	914	290	µg/kg	**yes**
Dioxins and Furans							
Dioxin TEQ	166	35,300	pg/g	42.0	14	pg/g	**yes**
Pesticides							
Aldrin	13 UT	691 J	µg/kg	1.99	8.0	µg/kg	no
cis-Chlordane	20 U	203 J	µg/kg	2.53	600	µg/kg	no
trans-Chlordane	9.8 U	445 NJ	µg/kg	1.45	600	µg/kg	no
Total chlordane	24.1 NJ	669 NJ	µg/kg	3.31	600	µg/kg	no
Sum DDD	190	3,040 NJ	µg/kg	24.7	180	µg/kg	no
Sum DDE	598	2,530 NJ	µg/kg	73.3	64	µg/kg	**yes**

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Table 2-13. Results of COPC Screen for Bald Eagle

COI	Exposure Concentrations			Dietary Dose			COPC?
	Max C_{prey} (ww)	Max C_{sed} (dw)	Unit	Max IR_{diet}	NOAEL TRV/ Eco SSL	Unit (bw/day)	
Sum DDT	295	12,500 NJ	µg/kg	43.2	30	µg/kg	**yes**
Total DDTs	764	16,200 NJ	µg/kg	102	180	µg/kg	no
Dieldrin	14 UT	356 J	µg/kg	1.90	66	µg/kg	no
alpha-Endosulfan	20 U	99 U	µg/kg	2.46	10,000	µg/kg	no
beta-Endosulfan	15 UT	235 NJ	µg/kg	1.95	10,000	µg/kg	no
Total Endosulfan	20 U	270	µg/kg	2.57	10,000	µg/kg	no
Endosulfan sulfate	12 UT	240 T	µg/kg	1.59	10,000	µg/kg	no
Endrin	31 UT	200 U	µg/kg	3.84	12	µg/kg	no
Endrin aldehyde	9.8 U	200 U	µg/kg	1.30	12	µg/kg	no
Endrin ketone	9.8 U	200 U	µg/kg	1.30	12	µg/kg	no
Heptachlor	13 UT	99 U	µg/kg	1.62	100	µg/kg	no
Heptachlor epoxide	9.8 U	99 U	µg/kg	1.24	100	µg/kg	no
alpha-HCH	9.8 U	99 U	µg/kg	1.24	1,600	µg/kg	no
beta-HCH	9.8 U	99 U	µg/kg	1.24	1,600	µg/kg	no
gamma-HCH	9.8 U	430	µg/kg	1.44	1,600	µg/kg	no
delta-HCH	9.8 U	99 U	µg/kg	1.24	1,600	µg/kg	no
Methoxychlor	17 JT	990 U	µg/kg	2.66	34,600	µg/kg	no
cis-Nonachlor	17 U	378 NJ	µg/kg	2.28	600	µg/kg	no
trans-Nonachlor	14 U	226 NJ	µg/kg	1.82	600	µg/kg	no
Oxychlordane	20 UT	130 U	µg/kg	2.48	600	µg/kg	no

[a] For butyltins, prey tissue was represented using clam tissue because TBT data in representative fish prey tissue were not available from the ERA dataset.

BEHP – bis(2-ethylhexyl) phthalate
bw – body weight
COI – chemical of interest
COPC – chemical of potential interest
dw – dry weight
Eco SSL – ecological soil screening level
HCH – hexachlorocyclohexane
IR – ingestion rate
J – estimated concentration
N – presumptive evidence of a compound
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyls
SVOC – semivolitile organic compound
T – value calculated or selected from multiple results
TEQ – toxic equivalent
TRV – toxicity reference value
U – not detected at given concentration
ww – wet weight

Bold indentifies when the maximum estimated dietary dose is greater than the NOAEL TRV or Eco SSL.

BZTO104(e)031293

Table 2-14. Results of COPC Screen for Osprey

COI	Exposure Concentrations			Dietary Dose			COPC?
	Max C_{prey} (ww)	Max C_{sed} (dw)	Unit	Max IR_{diet}	NOAEL TRV/ Eco SSL	Unit (bw/day)	
Metals							
Arsenic	0.42	75.6	mg/kg	0.171	2.24	mg/kg	no
Cadmium	0.108	46.2	mg/kg	0.0731	1.47	mg/kg	no
Chromium	2.77 T	774	mg/kg	1.43	2.66	mg/kg	no
Copper	1.42	1,080	mg/kg	1.48	4.05	mg/kg	no
Lead	0.303 J	1,950	mg/kg	2.19	1.63	mg/kg	**yes**
Mercury	0.494	4.84	mg/kg	0.109	0.01	mg/kg	**yes**
Nickel	1.37 J	594	mg/kg	0.936	77.4	mg/kg	no
Selenium	0.4	20	mg/kg	0.106	0.42	mg/kg	no
Thallium	0.0167	27	mg/kg	0.0330	0.48	mg/kg	no
Zinc	112	2,850	mg/kg	26.6	82	mg/kg	no
Butyltins							
Butyltin ion	97[a]	740 J	µg/kg	21.2	1,400	µg/kg	no
Dibutyltin ion	560[a]	2,700	µg/kg	121	1,400	µg/kg	no
Tetrabutyltin	3.4[a] U	1,000	µg/kg	1.80	1,400	µg/kg	no
Tributyltin ion	530[a]	47,000	µg/kg	163	1,400	µg/kg	no
PAHs							
Benzo(a)pyrene	33 UT	340,000	µg/kg	378	280	µg/kg	**yes**
Total PAHs	523	7,260,000	µg/kg	8,030	40,000	µg/kg	no
Phthalates							
BEHP	87,000 JT	440,000 J	µg/kg	18,800	1,450	µg/kg	**yes**
Butylbenzyl phthalate	660 U	2,800	µg/kg	142	1,450	µg/kg	no
Dibutyl phthalate	660 U	3,790 J	µg/kg	143	1,450	µg/kg	no
Diethyl phthalate	1,600 U	2,150 UJ	µg/kg	339	1,450	µg/kg	no
Dimethyl phthalate	330 U	2,150 UJ	µg/kg	71.7	1,450	µg/kg	no
Di-n-octyl phthalate	2,100 T	15,400 JT	µg/kg	458	1,450	µg/kg	no
SVOCs							
Hexachlorobenzene	33 UT	1,200 U	µg/kg	8.24	240	µg/kg	no
Hexachlorobutadiene	9.8 UJ	280 U	µg/kg	2.36	1,700	µg/kg	no
Phenols							
Pentachlorophenol	190 UT	8,410 J	µg/kg	49.1	22,000	µg/kg	no
PCBs							
PCB TEQ	82.7	5,610	pg/g	23.5	14	pg/g	**yes**
Total PCBs	7,460 J	30,800 J	µg/kg	1,600	290	µg/kg	**yes**
Dioxins and Furans							
Dioxin TEQ	22.5	35,300	pg/g	43.2	14	pg/g	**yes**

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Table 2-14. Results of COPC Screen for Osprey

COI	Exposure Concentrations			Dietary Dose			COPC?
	Max C_{prey} (ww)	Max C_{sed} (dw)	Unit	Max IR_{diet}	NOAEL TRV/ Eco SSL	Unit (bw/day)	
Pesticides							
Aldrin	13 UT	691 J	µg/kg	3.48	8.0	µg/kg	no
cis-Chlordane	27	203 J	µg/kg	5.89	600	µg/kg	no
trans-Chlordane	25	445 NJ	µg/kg	5.74	600	µg/kg	no
Total chlordane	73	669 NJ	µg/kg	16.1	600	µg/kg	no
Sum DDD	190	3,040 NJ	µg/kg	43.2	180	µg/kg	no
Sum DDE	598	2,530 NJ	µg/kg	128	64	µg/kg	**yes**
Sum DDT	295	12,500 NJ	µg/kg	75.6	30	µg/kg	**yes**
Total DDTs	764	16,200 NJ	µg/kg	178	180	µg/kg	no
Dieldrin	20 UT	356 J	µg/kg	4.59	66	µg/kg	no
alpha-Endosulfan	20 U	99 U	µg/kg	4.31	10,000	µg/kg	no
beta-Endosulfan	20 UT	235 NJ	µg/kg	4.46	10,000	µg/kg	no
Total endosulfan	28.4 J	270	µg/kg	6.26	10,000	µg/kg	no
Endosulfan sulfate	20 UT	240 T	µg/kg	4.46	10,000	µg/kg	no
Endrin	31 UT	200 U	µg/kg	6.73	12	µg/kg	no
Endrin aldehyde	20 UT	200 U	µg/kg	4.42	12	µg/kg	no
Endrin ketone	20 UT	200 U	µg/kg	4.42	12	µg/kg	no
Heptachlor	13 UT	99 U	µg/kg	2.84	100	µg/kg	no
Heptachlor epoxide	9.8 U	99 U	µg/kg	2.17	100	µg/kg	no
alpha-HCH	9.8 U	99 U	µg/kg	2.17	1,600	µg/kg	no
beta-HCH	9.8 U	99 U	µg/kg	2.17	1,600	µg/kg	no
gamma-HCH	9.8 U	430	µg/kg	2.53	1,600	µg/kg	no
delta-HCH	9.8 U	99 U	µg/kg	2.17	1,600	µg/kg	no
Methoxychlor	20 UT	990 U	µg/kg	5.28	34,600	µg/kg	no
cis-Nonachlor	17 U	378 NJ	µg/kg	3.98	600	µg/kg	no
trans-Nonachlor	15	226 NJ	µg/kg	3.40	600	µg/kg	no
Oxychlordane	32 UT	130 U	µg/kg	6.87	600	µg/kg	no

[a] For butyltins, prey tissue was represented using clam tissue because TBT data in representative fish prey tissue were not available from the ERA dataset.

BEHP – bis(2-ethylhexyl) phthalate
bw – body weight
COI – chemical of interest
COPC – chemical of potential interest
dw – dry weight
Eco SSL – ecological soil screening level
HCH – hexachlorocyclohexane
IR – ingestion rate
J – estimated concentration
N – presumptive evidence of a compound
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyls
SVOC – semivolitile organic compound
T – value calculated or selected from multiple results
TEQ – toxic equivalent
TRV – toxicity reference value
U – not detected at given concentration
ww – wet weight

Bold indentifies when the maximum estimated dietary dose is greater than the NOAEL TRV or Eco SSL.

Table 2-15. Results of COPC Screen for Mink

COI	Exposure Concentrations			Dietary Dose			COPC?
	Max C_{prey} (ww)	Max C_{sed} (dw)	Unit	Max IR_{diet}	NOAEL TRV/ Eco SSL	Unit (bw/day)	
Metals							
Antimony	0.02 J	32.1	mg/kg	0.129	0.059	mg/kg	**yes**
Arsenic	0.5 J	75.6	mg/kg	0.376	1.04	mg/kg	no
Cadmium	0.108	46.2	mg/kg	0.198	0.77	mg/kg	no
Chromium	2.77 T	774	mg/kg	3.47	5.66	mg/kg	no
Copper	17.6	1,080	mg/kg	7.04	5.82	mg/kg	**yes**
Lead	10.6	1,950	mg/kg	9.33	4.7	mg/kg	**yes**
Mercury	0.494	4.84	mg/kg	0.0981	0.02	mg/kg	**yes**
Nickel	1.37 J	594	mg/kg	2.54	8.4	mg/kg	no
Selenium	0.4	20	mg/kg	0.142	0.055	mg/kg	**yes**
Thallium	0.0167	27	mg/kg	0.108	0.74	mg/kg	no
Zinc	112	2,850	mg/kg	29.1	160	mg/kg	no
Butyltins							
Butyltin ion	97[a]	740 J	µg/kg	18.4	400	µg/kg	no
Dibutyltin ion	560[a]	2,700	µg/kg	100	3,800	µg/kg	no
Tetrabutyltin	3.4[a] U	1,000	µg/kg	4.46	400	µg/kg	no
Tributyltin ion	530[a]	47,000	µg/kg	269	400	µg/kg	no
PAHs							
2-Methylnaphthalene	73 UT	53,000	µg/kg	219	54,000	µg/kg	no
Benzo(a)pyrene	37 UT	340,000	µg/kg	1,340	2,000	µg/kg	no
Naphthalene	86 T	73,000	µg/kg	299	133,000	µg/kg	no
Total PAHs	731 J	7,260,000	µg/kg	28,500	2,000	µg/kg	**yes**
Phthalates							
BEHP	87,000 JT	440,000 J	µg/kg	15,700	44,000	µg/kg	no
Butylbenzyl phthalate	3,700 U	2,800	µg/kg	604	831,000	µg/kg	no
Dibutyl phthalate	660 U	3,790 J	µg/kg	121	16,000	µg/kg	no
Diethyl phthalate	1,600 U	2,150 UJ	µg/kg	265	1,860,000	µg/kg	no
Dimethyl phthalate	330 U	2,150 UJ	µg/kg	61.3	44,000	µg/kg	no
Di-n-octyl phthalate	2,100 T	15,400 JT	µg/kg	397	7,500,000	µg/kg	no
SVOCs							
Hexachlorobenzene	33 UT	1,200 U	µg/kg	9.98	26	µg/kg	no
Hexachlorobutadiene	33 UT	280 U	µg/kg	6.38	2,000	µg/kg	no
Hexachloroethane	120 U	1,500 J	µg/kg	25.1	100,000	µg/kg	no
Phenols							
Pentachlorophenol	420 U	8,410 J	µg/kg	100	4,000	µg/kg	no
Phenol	660 U	2,150 UJ	µg/kg	114	60,000	µg/kg	no
PCBs							
PCB TEQ (mammals)	43.7	324	pg/g	8.27	0.44	pg/g	**yes**
Total PCBs	7,460 J	30,800 J	µg/kg	1315	7.4	µg/kg	**yes**
Dioxins and Furans							
Dioxin TEQ (mammals)	38.6	16,600	pg/g	71.1	0.44	pg/g	**yes**

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Table 2-15. Results of COPC Screen for Mink

COI	Exposure Concentrations			Dietary Dose			COPC?
	Max C_{prey} (ww)	Max C_{sed} (dw)	Unit	Max IR_{diet}	NOAEL TRV/ Eco SSL	Unit (bw/day)	
Pesticides							
Aldrin	13 UT	691 J	µg/kg	4.79	830	µg/kg	no
cis-Chlordane	27	203 J	µg/kg	5.12	180	µg/kg	no
trans-Chlordane	25	445 NJ	µg/kg	5.74	180	µg/kg	no
Total chlordane	73	669 NJ	µg/kg	14.3	180	µg/kg	no
Total DDTs	3,060	16,200 NJ	µg/kg	553	260	µg/kg	**yes**
Dieldrin	24 JT	356 J	µg/kg	5.24	92	µg/kg	no
alpha-Endosulfan	20 U	99 U	µg/kg	3.59	840	µg/kg	no
beta-Endosulfan	20 UT	235 NJ	µg/kg	4.12	840	µg/kg	no
Total endosulfan	28.4 J	270	µg/kg	5.54	840	µg/kg	no
Endosulfan sulfate	20 UT	240 T	µg/kg	4.14	840	µg/kg	no
Endrin	31 UT	200 U	µg/kg	5.75	92	µg/kg	no
Endrin aldehyde	20 UT	200 U	µg/kg	3.99	92	µg/kg	no
Endrin ketone	20 UT	200 U	µg/kg	3.99	92	µg/kg	no
Heptachlor	13 UT	99 U	µg/kg	2.47	1,000	µg/kg	no
Heptachlor epoxide	9.8 U	99 U	µg/kg	1.96	1,000	µg/kg	no
alpha-HCH	9.8 U	99 U	µg/kg	1.96	6,100	µg/kg	no
beta-HCH	9.8 U	99 U	µg/kg	1.96	5,700	µg/kg	no
gamma-HCH	9.8 U	430	µg/kg	3.25	6,100	µg/kg	no
delta-HCH	9.8 U	99 U	µg/kg	1.96	6,100	µg/kg	no
Methoxychlor	20 UT	990 U	µg/kg	7.08	5,600	µg/kg	no
cis-Nonachlor	17 U	378 NJ	µg/kg	4.20	180	µg/kg	no
trans-Nonachlor	19 UT	226 NJ	µg/kg	3.93	2,500	µg/kg	no
Oxychlordane	32 UT	130 U	µg/kg	5.63	180	µg/kg	no

[a] For butyltins, prey tissue was represented using clam tissue because TBT data in representative fish or crayfish prey tissue were not available from the ERA dataset.

BEHP – bis(2-ethylhexyl) phthalate
bw – body weight
COI – chemical of interest
COPC – chemical of potential interest
dw – dry weight
Eco SSL – ecological soil screening level
HCH – hexachlorocyclohexane
IR – ingestion rate
J – estimated concentration
N – presumptive evidence of a compound
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyls
SVOC – semivolitile organic compound
T – value calculated or selected from multiple results
TEQ – toxic equivalent
TRV – toxicity reference value
U – not detected at given concentration
ww – wet weight

Bold indentifies when the maximum estimated dietary dose is greater than the NOAEL TRV or Eco SSL.

Table 2-16. Results of COPC Screen for River Otter

COI	Exposure Concentrations			Dietary Dose			COPC?
	Max C_{prey} (ww)	Max C_{sed} (dw)	Unit	Max IR_{diet}	NOAEL TRV/ Eco SSL	Unit (bw/day)	
Metals							
Antimony	0.02 J	32.1	mg/kg	0.0214	0.059	mg/kg	no
Arsenic	1.25	75.6	mg/kg	0.169	1.04	mg/kg	no
Cadmium	0.218	46.2	mg/kg	0.0494	0.77	mg/kg	no
Chromium	2.77 T	774	mg/kg	0.740	5.66	mg/kg	no
Copper	17.6	1,080	mg/kg	2.39	5.82	mg/kg	no
Lead	1.3 JT	1,950	mg/kg	1.31	4.7	mg/kg	no
Mercury	0.114	4.84	mg/kg	0.0142	0.02	mg/kg	no
Nickel	1.37 J	594	mg/kg	0.494	8.4	mg/kg	no
Selenium	0.4	20	mg/kg	0.0515	0.055	mg/kg	no
Thallium	0.0167	27	mg/kg	0.0180	0.74	mg/kg	no
Zinc	112	2,850	mg/kg	12.8	160	mg/kg	no
Butyltins							
Butyltin ion	97	740 J	µg/kg	10.0	400	µg/kg	no
Dibutyltin ion	560	2,700	µg/kg	56.8	3,800	µg/kg	no
Tetrabutyltin	3.4 U	1,000	µg/kg	0.939	400	µg/kg	no
Tributyltin ion	530	47,000	µg/kg	80.6	400	µg/kg	no
PAHs							
2-Methylnaphthalene	73 UT	53,000	µg/kg	39.2	54,000	µg/kg	no
Benzo(a)pyrene	490	340,000	µg/kg	254	2,000	µg/kg	no
Naphthalene	86 T	73,000	µg/kg	52.6	133,000	µg/kg	no
Total PAHs	4,980	7,260,000	µg/kg	4,880	2,000	µg/kg	**yes**
Phthalates							
BEHP	87,000 JT	440,000 J	µg/kg	8,840	44,000	µg/kg	no
Butylbenzyl phthalate	660 U	2,800	µg/kg	66.7	831,000	µg/kg	no
Dibutyl phthalate	1,300	3,790 J	µg/kg	130	16,000	µg/kg	no
diethyl phthalate	1,600 U	2,150 UJ	µg/kg	159	1,860,000	µg/kg	no
Dimethyl phthalate	330 U	2,150 UJ	µg/kg	33.8	44,000	µg/kg	no
Di-n-octyl phthalate	2,100 T	15,400 JT	µg/kg	216	7,500,000	µg/kg	no
SVOCs							
Hexachlorobenzene	33 UT	1,200 U	µg/kg	3.98	26	µg/kg	no
Hexachlorobutadiene	33 UT	280 U	µg/kg	3.42	2,000	µg/kg	no
Hexachloroethane	37 UT	1,500 J	µg/kg	4.55	100,000	µg/kg	no
Phenols							
Pentachlorophenol	190 UT	8,410 J	µg/kg	23.8	4,000	µg/kg	no
Phenol	2,600 T	2,150 UJ	µg/kg	258	60,000	µg/kg	no
PCBs							
PCB TEQ (mammals)	43.7	324	pg/g	4.50	0.44	pg/g	**yes**
Total PCBs	7,460 J	30,800 J	µg/kg	754	7.4	µg/kg	**yes**
Dioxins and Furans							
Dioxin TEQ (mammals)	38.6	16,600	pg/g	13.8	0.44	pg/g	**yes**
Pesticides							
Aldrin	13 UT	691 J	µg/kg	1.70	830	µg/kg	no

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Table 2-16. Results of COPC Screen for River Otter

COI	Exposure Concentrations			Dietary Dose			COPC?
	Max C_{prey} (ww)	Max C_{sed} (dw)	Unit	Max IR_{diet}	NOAEL TRV/ Eco SSL	Unit (bw/day)	
cis-Chlordane	20 U	203 J	µg/kg	2.09	180	µg/kg	no
trans-Chlordane	9.6 U	445 NJ	µg/kg	1.22	180	µg/kg	no
Total chlordane	32 U	669 NJ	µg/kg	3.56	180	µg/kg	no
Total DDTs	3,060	16,200 NJ	µg/kg	311	260	µg/kg	**yes**
alpha-Endosulfan	20 U	99 U	µg/kg	2.03	840	µg/kg	no
beta-Endosulfan	20 UT	235 NJ	µg/kg	2.11	840	µg/kg	no
Total endosulfan	28.4 J	270	µg/kg	2.96	840	µg/kg	no
Dieldrin	24 JT	356 J	µg/kg	2.58	92	µg/kg	no
Endosulfan sulfate	20 UT	240 T	µg/kg	2.12	840	µg/kg	no
Endrin	31 UT	200 U	µg/kg	3.18	92	µg/kg	no
Endrin aldehyde	20 UT	200 U	µg/kg	2.09	92	µg/kg	no
Endrin ketone	20 UT	200 U	µg/kg	2.09	92	µg/kg	no
Heptachlor	13 UT	99 U	µg/kg	1.34	1,000	µg/kg	no
Heptachlor epoxide	9.6 U	99 U	µg/kg	1.01	1,000	µg/kg	no
alpha-HCH	9.6 U	99 U	µg/kg	1.01	6,100	µg/kg	no
beta-HCH	9.6 U	99 U	µg/kg	1.01	5,700	µg/kg	no
gamma-HCH	9.6 UT	430	µg/kg	1.21	6,100	µg/kg	no
delta-HCH	9.6 U	99 U	µg/kg	1.01	6,100	µg/kg	no
Methoxychlor	20 UT	990 U	µg/kg	2.57	5,600	µg/kg	no
cis-Nonachlor	17 U	378 NJ	µg/kg	1.90	180	µg/kg	no
trans-Nonachlor	11 UT	226 NJ	µg/kg	1.22	2,500	µg/kg	no
Oxychlordane	32 UT	130 U	µg/kg	3.23	180	µg/kg	no

BEHP – bis(2-ethylhexyl) phthalate
bw – body weight
COI – chemical of interest
COPC – chemical of potential interest
dw – dry weight
Eco SSL – ecological soil screening level
HCH – hexachlorocyclohexane
IR – ingestion rate
J – estimated concentration
N – presumptive evidence of a compound
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyls
SVOC –semivolitile organic compound
T – value calculated or selected from multiple results
TEQ – toxic equivalent
TRV – toxicity reference value
U – not detected at given concentration
ww –wet weight

Bold indentifies when the maximum estimated dietary dose is greater than the NOAEL TRV or Eco SSL.

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Table 2-17. Receptor/COPC Pairs Evaluated for Wildlife Receptors

COPC	Wildlife Receptor					
	Spotted Sandpiper	**Hooded Merganser**	**Bald Eagle**	**Osprey**	**Mink**	**River Otter**
Metals						
Antimony					X	
Arsenic	X					
Cadmium	X					
Chromium	X					
Copper	X	X			X	
Lead	X	X	X	X	X	
Mercury	X	X	X	X	X	
Selenium	X				X	
Thallium	X					
Zinc	X					
Butyltins						
Butyltin ion	X					
Tributyltin ion	X					
PAHs						
Benzo(a)pyrene	X	X		X		
Total PAHs	X				X	X
Phthalates						
BEHP	X	X	X	X		
Dibutyl phthalate	X					
PCBs						
PCB TEQ	X	X	X	X	X	X
Total PCBs	X	X	X	X	X	X
Dioxins						
Dioxin TEQ	X	X	X	X	X	X
Pesticides						
Aldrin	X					
Sum DDD	X					
Sum DDE	X	X	X	X		
Sum DDT	X	X	X	X		
Total DDTs	X	X			X	X

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

Table 3-1. Bird Egg NOAEL TRVs Used for Screening to Determine COPCs

COI	Species	NOAEL TRV (µg/g ww)	Endpoint Effect	Source
Mercury	bald eagle	0.50	5-year productivity	Wiemeyer et al. (1984)
PCB TEQ	chicken	0.00000742	embryo abnormalities	Henshel et al. (1993)
Total PCBs	chicken	0.41	egg production/egg hatchability	Scott et al. (1975)
Dioxin TEQ	chicken	0.00000742	embryo abnormalities	Henshel et al. (1993)
Sum DDE	bald eagle	1.3	5-year productivity	Wiemeyer et al. (1984)

COI – chemical of interest
COPC –chemical of potential concern
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
TRV – toxicity reference value
ww – wet weight

Table 3-2. Results of the COPC Screen for Bald Eagle and Osprey Using the Bird Egg Approach

COI	BMF (unitless)	Exposure Concentration		Egg Concentration			COPC?
		Maximum C_{prey}	Unit (ww)	Maximum EPC_{egg}	NOAEL TRV	Unit (ww)	
Dioxin TEQ	10	166	ng/kg	1,660	7.42	ng/kg	**yes**
PCB TEQ	10	117	ng/kg	1,170	7.42	ng/kg	**yes**
Total PCBs	11	7,460 J	µg/kg	82,100	410	µg/kg	**yes**
Mercury	3	0.494	mg/kg	1.48	0.5	mg/kg	**yes**
Sum DDE	87	657	µg/kg	57,200	1,300	µg/kg	**yes**

BMF – biomagnifications factor
COI – chemical of interest
COPC – chemical of potential concern
EPC – exposure point concentration
J – estimated concentration
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
TRV – toxicity reference value
ww – wet weight
Bold identifies where maximum estimated egg concentration is greater than the NOAEL TRV.

APPENDIX G: ECOLOGICAL RISK ASSESSMENT

Attachment G7

Screening Assessment for Amphibians and Reptiles

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1.0 INITIAL SCREENING PROCESS

This attachment presents the screening assessment that was conducted to identify chemicals of potential concern (COPCs) for amphibian and reptile receptors in the Lower Willamette River (LWR). In the initial screening process, one pathway of exposure and effect was evaluated; surface water data were compared to ambient water quality criteria (AWQC) or TRVs, to evaluate risks to amphibians. Risks to reptiles were not evaluated as stated in Appendix B of the Programmatic Work Plan (Integral et al. 2004). Surface water is the only pathway that is considered complete and significant for amphibians, and for which relevant toxicity data were available. The results of the surface water screen are presented in Section 2.0 and the further screening evaluation of surface water Round 2 COPCs and screening conclusions for amphibians are presented in Section 3.0.

2.0 SURFACE WATER SCREEN

This section presents the initial screening process for surface water assessment including screening of surface water data and Round 2 COPC identification.

2.1 SELECTION OF COIS

Chemicals of interest (COIs) for amphibians were defined as all surface water chemicals detected in the quiescent areas of the Study Area, excluding crustal elements (i.e., aluminum; see Section 2.0 of Appendix G). The list of COIs for amphibians is presented in Table 2-1.

2.2 IDENTIFICATION OF ROUND 2 COPCS

To identify Round 2 COPCs for amphibians, the maximum detected surface water concentration of each COI was compared to the respective chronic Eco SL. If the maximum exposure concentration was greater than the Eco SL, the chemical was identified as a Round 2 COPC. This screening approach is consistent with EPA guidance (EPA 2006a).

A review of water quality regulatory benchmarks and literature-based thresholds was conducted to develop the Ecological Screening Levels (Eco SLs) for chemicals in water. Eco SLs were developed for all amphibian surface water COIs except individual dioxins and furans other than 2,3,7,8-TCDD because no data were available. Eco SLs, presented in Attachment G3, were developed by LWG and revised based on EPA's comments (EPA 2006a). LWG adopted EPA's proposed hierarchy for Eco SL selection (LWG 2006) and in accordance with EPA's comments, a use factor (UF) of 50 was used to calculate a chronic screening value from an acute screening value when no chronic data were available. There is high uncertainty in the use of a UF of 50, and LWG recommends the use of a range of UFs for the Baseline Environmental Risk Assessment (BERA). The Eco SLs are considered to be protective of all aquatic receptors, including benthic invertebrates, fish, and amphibians.

Table 2-2 presents the maximum concentrations, the corresponding chronic Eco SL, and the Round 2 COPC determination for each amphibian COI. The following six COIs were identified as surface water Round 2 COPCs for amphibians:

- Zinc (dissolved)
- 4-Chloro-3-methylphenol
- Total PCBs
- 2,4′-DDT
- 4,4′-DDT
- Total DDTs

2.3 FURTHER SCREENING EVALUATION

Further screening of Round 2 surface water COPCs was conducted for amphibians. First, a general habitat assessment of the surface water sampling locations for amphibians was conducted to provide a general understanding of the overall exposure to amphibians across the Study Area and across seasonal changes. Second, a comparison of the surface water Eco SLs to amphibian-specific toxicity data was conducted.

2.3.1 Potential Habitat and Amphibian Exposure Evaluation

The exposure area for amphibians consisted of discrete quiescent backwater areas (Table 2-3) where surface water samples were collected. Sampling methods used for the collection of surface water in the identified amphibian habitat areas were either peristaltic or XAD or both. Some uncertainty may be associated with the data analyzed from the peristaltic samples because these samples were collected over a short period of time and may overestimate exposure concentrations for and risks to amphibians. Details on surface water sampling methods are described in Section 2.0 of Appendix G. Table 2-4 presents the chemical concentrations for the six Round 2 COPCs.

Habitat at the 14 amphibian surface water sampling locations is similar; all are quiet backwater areas with shallow water and emergent vegetation. Based on observations made during the 2002 amphibian reconnaissance survey and multiple sampling events, the habitat available to amphibians in these areas was divided into two general types: low-sloping beaches and steeper, riprapped or rocky banks, as detailed below:

- **Low-sloping beaches**—Generally have shallower water, aquatic vegetation (both emergent and submersed), and/or large woody debris. This type of habitat is present at W001, W0003, W013, W017, and W021.
- **Steeper, riprapped or rocky banks**—Generally are adjacent to deeper water and may also support emergent vegetation or shrubs. This type of habitat can be found at W004, W006, W007, W008, W009, W019, and W022.

All of the amphibian habitat areas associated with the surface water sampling locations between RM 3.5 and RM 9.2 were evaluated during the 2002 amphibian/ reptile reconnaissance survey (Integral et al. 2004), with the exception of the area in the at W002 and W018. Amphibians or evidence of amphibians (e.g., frog calls or egg masses) were noted at both habitat types. Two northern red-legged frogs were observed at W003, frog calls were heard at W008 (species not identified) and W021 (Pacific tree frog), and egg masses (species not identified) were observed at W004 and W007. No information is available on the types of habitat present in the vicinity of W002 and W018 because these locations were not evaluated during the survey.

Round 2 sampling activities at the 14 amphibian sampling locations involved three surface water sampling events (i.e., November 2004, March 2005, and July 2005) to capture the seasonality of the water flow for the LWR (Integral 2004). The November 2004 event was conducted during mid- to late fall to capture potentially elevated chemical concentrations in the river from rainfall runoff. The March 2005 event was selected by EPA to coincide with the early exposure period for amphibian egg masses. This sampling event likely represents the most ecologically relevant exposure period for amphibians. Amphibians are not likely to use aquatic habitat and thus be exposed to surface water during the late fall (as represented by the November sampling event) or low flow season (as represented by the July sampling event). Therefore, exceedances from data collected during the November and July sampling events may overestimate risks to amphibians. The July 2005 event was timed to coincide with low-flow conditions, when any effects of groundwater discharge to the water column would be most pronounced. Unfortunately, river flow was atypical and represented low-flow conditions during all three sampling events. Additional surface water sampling will be conducted as part of Round 3A sampling. Some locations will be adjacent to the quiescent areas, and data collected from these locations may be used to evaluate risks to amphibians.

2.3.2 Evaluation of Amphibian Toxicity Data

Eco SLs were selected based on the proposed hierarchy detailed in Attachment G3. Table 2-5 presents the chronic and acute Eco SLs for amphibian Round 2 COPCs. For all amphibian Round 2 COPCs except 4-chloro-3-methylphenol (which was based on water quality criteria protective of aquatic life), Eco SLs were based on toxicity studies that examined effects on invertebrate and fish species. The relevancy of these values to amphibian species is unknown, and therefore, amphibian-specific toxicity data were evaluated and presented, where data were available:

2.3.2.1 Zinc

The Eco SLs for zinc, which are based on AWQC, are hardness-dependent and were adjusted from the AWQC values using EPA-provided equations (EPA 2006b) and a hardness of 25 mg/L calcium carbonate (as estimated for the LWR). Criteria are for the dissolved fraction of zinc, which is the bioavailable fraction.

Amphibian-specific toxicity data on zinc in water were available and reviewed in the original Eco SL table submitted to EPA (Windward 2005). The recommended acute and chronic screening levels for amphibians were 0.005 and 0.0005 mg/L. These data were derived from a 7-day LC50 study (Birge et al. 1979) to which uncertainty factors of 2 and 20 were applied to derive the acute and chronic screening levels, respectively. These amphibian-specific values are lower than the selected Eco SLs; therefore, the Eco SLs may underestimate risks to amphibians. However, toxicological data for zinc and amphibians is extremely limited, and the use of uncertainty factors is uncertain (see Section 2.4).

2.3.2.2 PCBs

The acute Eco SL for total PCBs was based on the value (or benchmark) developed by ODEQ and is the same as the Tier II value presented in Suter and Tsao (1996). The acute Eco SL was calculated for AWQC, although it has never been adopted. The chronic Eco SL was based on ODEQ based on partial life-cycle tests conducted with three invertebrate and two fish species. Suter and Tsao cited the same chronic value for total PCBs and noted that it was based on fish residual values.

No amphibian-specific toxicity data on total PCBs were available. However, toxicity data on the exposure of amphibians to Aroclor 1254 in water were available and reviewed in the original Eco SL table submitted to EPA (Windward 2005). The recommended acute and chronic screening levels for amphibians were 63 and 6.3 µg/L, respectively. These values were derived from a 96-hour study in which 70% mortality was observed (Zhou et al. 2004) to which uncertainty factors of 4 and 40 were applied to derive the acute and chronic screening levels, respectively. The chronic Eco SL is two orders of magnitude lower than the amphibian-specific chronic value. Therefore, the Eco SLs may overestimate risks to amphibians. However, the toxicological data for PCBs and amphibians is extremely limited, and the use of uncertainty factors is uncertain (see Section 2.4).

2.3.2.3 DDTs

The Eco SLs for 2,4′-DDD, 4,4′-DDD, 2,4′-DDT, 4,4′-DDT, and total DDTs are based on the acute and chronic AWQC for 4,4′-DDT. No amphibian-specific toxicity data on 2,4′-DDT and 4,4′-DDT were available. However, toxicity data on the exposure of amphibians to total DDT in water were available and reviewed in the original Eco SL table submitted to EPA (Windward 2005). The recommended acute and chronic screening levels for amphibians were 2,360 and 236 µg/L, respectively. These data were derived from a 96-hour developmental study in which growth was reduced (Saka 2004) to which uncertainty factors of 2 and 20 were applied to derive the acute and chronic screening levels, respectively. The chronic Eco SL is three orders of magnitude lower than the amphibian-specific chronic value. Therefore, the Eco SLs may overestimate risks to amphibians. However, the toxicological data for total DDTs and amphibians is extremely limited, and the use of uncertainty factors is uncertain (see Section 2.4).

2.3.2.4 4-Chloro-3-methylphenol

The acute Eco SLs for 4-chloro-3-methylphenol was obtained from ODEQ's Water Quality Guidance Values Summary (Table 33), which can be used in applying Oregon's Narrative Toxics Criteria (Oregon Administrative Rules [OAR] 340-041-0033(1)) to waters of the state in order to protect aquatic life (ODEQ 2001). The source for this acute criterion is uncertain. In accordance with EPA's comments (EPA 2006a), a UF of 50 was used to extrapolate a chronic screening value from the acute Eco SL because no chronic data were available; however, there is high uncertainty in the use of a UF of 50. No amphibian-specific toxicity data on 4-chloro-3-methylphenol and amphibians were available (Windward 2005). The Eco SLs may

over or underestimate risks to amphibians because no toxicological data on amphibian species are available for comparison.

3.0 SCREENING ASSESSMENT CONCLUSIONS

The following section presents that uncertainties and screening conclusions for amphibians.

3.1 UNCERTAINTIES

There are several uncertainties associated with the amphibian surface water screening assessment, including the following (also summarized in Table 2-6):

- **The use of Eco SLs for amphibians**—Amphibian-specific toxicity data for total PCBs and total DDTs were several orders of magnitude higher than the chronic Eco SLs and are more relevant to evaluate the potential risks to amphibians. In addition, the total DDT value was used as a surrogate for 2,4′-DDT and 4,4′-DDT. None of these chemicals were detected at concentrations greater than the amphibian-specific toxicity criteria.
- **Surface water sampling methods**—All of the Round 2 COPC concentrations that are greater than the chronic Eco SLs are based on the surface water results from the peristaltic method. Locations W013 and W018, which were both sampled using the XAD method, do not have concentrations greater than Eco SLs. Some uncertainty may be associated with the data analyzed from the peristaltic samples because these samples were collected over a short period of time and may overestimate exposure concentrations for and risks to amphibians.
- **Limited data on LWR amphibian population and distribution**—The distribution and population of amphibians at the Study Area is uncertain, and it is unknown whether amphibians are present in or use all habitat areas where surface water sampling stations were located.
- **Seasonal occurrence of amphibian habitat**—The March 2005 sampling event likely represents the most ecologically relevant exposure period for amphibians. Amphibians are not likely to use aquatic habitat and thus be exposed to surface water during the late fall (as represented by the November sampling event) or low flow season (as represented by the July sampling event).
- **The use of NJ-qualified data**—Surface water concentrations of 2,4′-DDT and total DDTs at W001 were greater than Eco SLs, but were each based on a single NJ-qualified result. 2,4′-DDT is the only isomer that was detected in this sample. NJ-qualification indicates "the presence of an analyte that has been 'tentatively identified,' and the associated numerical value represents its approximate concentration" (EPA 1999). The qualification indicates that the analyst believed that the result was due to analytical interference from a chemical other than

the target analyte. The NJ-qualified result is biased high and will result in an overestimation of risk.

3.2 AMPHIBIAN SCREENING CONCLUSIONS

Table 2-6 presents a summary of surface water locations where Round 2 COPCs are greater than Eco SLs. Surface water concentrations were greater than the chronic Eco SL at only one surface water location within the Study Area for total PCBs, zinc, 2,4′ DDT, 4,4′-DDT, and 4-chloro-3-methylphenol. Thus, these chemicals would not pose a risk to amphibians at the population level. Furthermore, amphibian-specific toxicity data for total PCBs and total DDTs[1] were several orders of magnitude higher than the Eco SLs and are more relevant to evaluate the potential risks to amphibians. None of these chemicals were detected at concentrations greater than the amphibian-specific toxicity criteria. Therefore, based on this screening assessment, none of the six Round 2 COPCs for amphibians were identified as iCOCs. Amphibians and reptiles were not further evaluated.

[1] The total DDT value was used as a surrogate for 2,4′-DDT and 4,4′-DDT.

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TABLES

Table 2-1. COIs for Amphibians

Chemicals of Interest	
Metals	
Antimony	Nickel
Arsenic	Selenium
Cadmium	Silver
Chromium	Thallium
Copper	Zinc
Lead	
Butyltins	
Butyltin ion	Tributyltin ion
Dibutyltin ion	
PAHs	
2-Methylnaphthalene	Chrysene
Acenaphthene	Dibenzo(a,h)anthracene
Acenaphthylene	Fluoranthene
Anthracene	Fluorene
Benzo(a)anthracene	Indeno(1,2,3-cd)pyrene
Benzo(a)pyrene	Naphthalene
Benzo(b)fluoranthene	Phenanthrene
Benzo(b+j)fluoranthene	Pyrene
Benzo(g,h,i)perylene	
Benzo(k)fluoranthene	
Phthalates	
BEHP	Diethyl phthalate
Butylbenzyl phthalate	Dimethyl phthalate
Dibutyl phthalate	Di-n-octyl phthalate
SVOCs	
1,4-Dichlorobenzene	Hexachlorobenzene
Benzoic acid	Hexachlorobutadiene
Carbazole	Isophorone
Phenols	
4-Chloro-3-methylphenol	Phenol
PCBs	
Aroclor 1242	Aroclor 1260
Aroclor 1248	Total PCBs
Aroclor 1254	
Dioxins and Furans	
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin	1,2,3,6,7,8-Hexachlorodibenzofuran
2,3,7,8-Tetrachlorodibenzofuran	1,2,3,7,8,9-Hexachlorodibenzofuran
1,2,3,7,8-Pentachlorodibenzo-*p*-dioxin	2,3,4,6,7,8-Hexachlorodibenzofuran
2,3,4,7,8-Pentachlorodibenzofuran	1,2,3,4,6,7,8-Heptachlorodibenzo-*p*-dioxin

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Table 2-1. COIs for Amphibians

Chemicals of Interest	
1,2,3,7,8-Pentachlorodibenzofuran	1,2,3,4,6,7,8-Heptachlorodibenzofuran
1,2,3,4,7,8-Hexachlorodibenzo-*p*-dioxin	1,2,3,4,7,8,9-Heptachlorodibenzofuran
1,2,3,6,7,8-Hexachlorodibenzo-*p*-dioxin	Octachlorodibenzo-*p*-dioxin
1,2,3,7,8,9-Hexachlorodibenzo-*p*-dioxin	Octachlorodibenzofuran
1,2,3,4,7,8-Hexachlorodibenzofuran	
Herbicides	
Dalapon	
Pesticides	
Aldrin	Endosulfan sulfate
cis-Chlordane	Endrin
trans-Chlordane	Endrin ketone
Total chlordane	Heptachlor
2,4′-DDD	Heptachlor epoxide
2,4′-DDE	alpha-HCH
2,4′-DDT	beta-HCH
4,4′-DDD	gamma-HCH
4,4′-DDE	delta-HCH
4,4′-DDT	Methoxychlor
Total DDTs	cis-Nonachlor
Dieldrin	trans-Nonachlor
alpha-Endosulfan	Oxychlordane
beta-Endosulfan	

BEHP – bis(2-ethylhexyl) phthalate
COI – chemical of interest
HCH – hexachlorocyclohexane
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound

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Table 2-2. Round 2 COPCs Identified in Surface Water for Amphibians

COI	Units	Maximum Surface Water Concentration	Chronic Eco SL	Round 2 COPC?[a]
Metals[b]				
Antimony (total)	mg/L	0.000125 T	0.03	no
Arsenic (dissolved)	mg/L	0.00064	0.15	no
Cadmium (dissolved)	mg/L	0.00003	0.00009	no
Chromium (dissolved)	mg/L	0.00033	0.0238	no
Copper (dissolved)	mg/L	0.00164 J	0.00274	no
Lead (dissolved)	mg/L	0.000152	0.00054	no
Nickel (dissolved)	mg/L	0.00111	0.0161	no
Selenium (total)	mg/L	0.0009 J	0.005	no
Silver (dissolved)	mg/L	0.000061	0.0001	no
Thallium (total)	mg/L	0.000013 J	0.04	no
Zinc (dissolved)	mg/L	0.0419	0.0365	**yes**
Butyltins				
Butyltin ion	µg/L	0.02 J	0.072	no
Dibutyltin ion	µg/L	0.0073 J	0.072	no
Tributyltin ion	µg/L	0.0023 J	0.072	no
PAHs				
2-Methylnaphthalene	µg/L	0.062	2.1	no
Acenaphthene	µg/L	0.0262	23	no
Acenaphthylene	µg/L	0.012 J	306.9	no
Anthracene	µg/L	0.011 J	0.73	no
Benzo(a)anthracene	µg/L	0.012 J	0.027	no
Benzo(a)pyrene	µg/L	0.0075 J	0.014	no
Benzo(b)fluoranthene	µg/L	0.009 J	0.6774	no
Benzo(b+j)fluoranthene	µg/L	0.0023505 J	0.6415	no
Benzo(g,h,i)perylene	µg/L	0.00185	0.4391	no
Benzo(k)fluoranthene	µg/L	0.008 J	0.6415	no
Chrysene	µg/L	0.0089 J	2.042	no
Dibenzo(a,h)anthracene	µg/L	0.0072 J	0.2825	no
Fluoranthene	µg/L	0.082	6.16	no
Fluorene	µg/L	0.021	3.9	no
Indeno(1,2,3-cd)pyrene	µg/L	0.0057 J	0.275	no
Naphthalene	µg/L	0.0169	12	no
Phenanthrene	µg/L	0.096	6.3	no
Pyrene	µg/L	0.047	10.11	no
Phthalates				
BEHP	µg/L	0.025 J	3	no

Table 2-2. Round 2 COPCs Identified in Surface Water for Amphibians

COI	Units	Maximum Surface Water Concentration	Chronic Eco SL	Round 2 COPC?[a]
Butylbenzyl phthalate	µg/L	0.027 J	3	no
Dibutyl phthalate	µg/L	0.00217 J	3	no
Diethyl phthalate	µg/L	0.00233	3	no
Dimethyl phthalate	µg/L	0.0000344 J	3	no
Di-n-octyl phthalate	µg/L	0.000142 J	3	no
SVOCs				
1,4-Dichlorobenzene	µg/L	0.019 J	15	no
Benzoic acid	µg/L	2.2 J	42	no
Carbazole	µg/L	0.03 J	18.6	no
Hexachlorobenzene	µg/L	0.00701	3.68	no
Hexachlorobutadiene	µg/L	0.00256	9.3	no
Isophorone	µg/L	0.018 J	130	no
Phenols				
4-Chloro-3-methylphenol	µg/L	0.65	0.6	**yes**
Phenol	µg/L	0.17	110	no
PCBs				
Aroclor 1242	µg/L	0.00007044 NJ	0.053	no
Aroclor 1248	µg/L	0.00705 J	0.081	no
Aroclor 1254	µg/L	0.00657	0.033	no
Aroclor 1260	µg/L	0.008232 NJ	94	no
Total PCBs	µg/L	0.01618	0.014	**yes**
Dioxins and Furans				
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin[c]	pg/L	0.26	100	no
Pesticides				
Aldrin	µg/L	0.000002143 J	0.3	no
cis-Chlordane	µg/L	0.00104 J	0.0043	no
trans-Chlordane	µg/L	0.00108 J	0.0043	no
Total chlordane	µg/L	0.00293 J	0.0043	no
2,4′-DDD	µg/L	0.0000468	0.001	no
2,4′-DDE	µg/L	0.00000317 J	0.001	no
2,4′-DDT	µg/L	0.0187 NJ	0.001	**yes**
4,4′-DDD	µg/L	0.0001072	0.001	no
4,4′-DDE	µg/L	0.0000609	0.001	no
4,4′-DDT	µg/L	0.00115	0.001	**yes**
Total DDTs	µg/L	0.0199 NJ	0.001	**yes**
Dieldrin	µg/L	0.00004427 J	0.056	no
alpha-Endosulfan	µg/L	0.000569	0.056	no
beta-Endosulfan	µg/L	0.0000331	0.056	no

Table 2-2. Round 2 COPCs Identified in Surface Water for Amphibians

COI	Units	Maximum Surface Water Concentration	Chronic Eco SL	Round 2 COPC?[a]
Endosulfan sulfate	µg/L	0.00077 J	0.056	no
Endrin	µg/L	0.000000787 J	0.036	no
Endrin ketone	µg/L	0.0000011 J	0.036	no
Heptachlor	µg/L	0.000000141 J	0.0038	no
Heptachlor epoxide	µg/L	0.000004911 J	0.0038	no
alpha-HCH	µg/L	0.000076215 J	0.08	no
beta-HCH	µg/L	0.00000832 J	0.08	no
delta-HCH	µg/L	0.00167 NJ	0.08	no
gamma-HCH	µg/L	0.000768	0.08	no
Methoxychlor	µg/L	0.0111	0.03	no
cis-Nonachlor	µg/L	0.000002263 J	0.0043	no
trans-Nonachlor	µg/L	0.00000929 J	0.0043	no
Oxychlordane	µg/L	0.00000267 J	0.0043	no
Herbicides				
Dalapon	µg/L	0.23	2	no

[a] Round 2 COPCs were identified where the maximum detected surface water concentration was greater than the chronic Eco SL.

[b] Concentrations of metals were evaluated as total or dissolved concentrations, depending on the Eco SL.

[c] 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin was evaluated as a representative for all dioxins and furans.

BEHP – bis(2-ethylhexyl) phthalate
COI – chemical of interest
COPC – chemical of potential concern
Eco SL – Ecological Screening Level
EF – exceedance factor
HCH – hexachlorocyclohexane
J – estimated concentration
N – presumptive evidence of a compound
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
Bold identifies where the maximum detected surface water concentration was greater than the chronic Eco SL.

Table 2-3. Summary of Amphibian Exposure Areas

Sampling Location	General Vicinity	Habitat Type
W001	on east bank, downstream of RM 2, near Oregon Steel Mills	Low-sloping beach, shallow water, aquatic vegetation, and/or large woody debris.
W002[a]	Sauvie Island	Area not visited during survey; habitat is unknown.
W003	near Multnomah Channel	Upland natural bank drops off sharply to low-sloping vegetated beach and shallow water. Large patches of emergent vegetation and large woody debris.
W004[a]	International Slip	Steep, highly vegetated natural bank that drops off quickly into deep water.
W006	near Kinder Morgan Linnton	Steeply sloped riprap-covered bank that drops off quickly to deep waters. Some large woody debris.
W007	Terminal 4/Slip 1	Steep vegetated bank at end of slip that drops off quickly into deep waters. Emergent vegetation present.
W008	Terminal 4/Slip3	Steep rocky bank that is sparsely vegetated and drops off quickly into very deep water. Abundant large woody debris.
W009	downstream from St. Johns Bridge	Steeply sloped riprap-covered bank that drops off quickly to deep waters. Abundant large woody debris.
W013[a]	Willamette Cove	Low-sloping sandy beach with some vegetation, woody debris, and concrete pilings.
W017	Saltzman Creek, Willbridge	Low-sloping sandy beach adjacent to backwater marsh. Some large woody debris. Few emergent plants are located on the fringe of the backwater area.
W018	near Portland Shipyard	Area not visited during survey; habitat is unknown.
W019	near Gunderson	Narrow rocky outcrop extending out from steep bank. Narrow strip of shallow water drops off quickly into deeper water. Few emergent plants.
W021	Swan Island Lagoon	Low-sloping sandy beach with some vegetation and woody debris. Emergent vegetation present in shallow water.
W022	Terminal 2	Steep bank, deep water, and some emergent vegetation.

[a] Field replicate samples were collected at these locations.

RM – river mile

Table 2-4. Chemical Concentrations for Round 2 COPCs in Surface Water for Amphibians

Sampling Location	Sampling Event[a]	Sampling Method	Zinc (dissolved; mg/L)	Total PCBs (µg/L)	2,4′-DDT (µg/L)	4,4′-DDT (µg/L)	Total DDTs (µg/L)	4-Chloro-3-methylphenol (µg/L)
W001	1	peristaltic	0.0014	0.0025 UJ	0.000472 UJ	0.000472 UJ	0.000472 UJ	0.029 U
	2		0.0014 T	0.0025 U	0.0187 NJ	0.00049 U	0.0199 NJ	0.032 U
	3		0.00121 UJ	0.0088 J	0.000481 UJ	0.000481 UJ	0.000481 UJ	0.048 U
W002[b]	1	peristaltic	0.0023	NA	0.000481 UJ	0.000481 UJ	0.000481 UJ	0.029 U
	2		0.0012	0.00255 U	0.000495 U	0.000495 U	0.000495 U	0.41 U
	2 (FR)		0.0011	NA	NA	NA	NA	0.68 U
	3		0.00117 UJ	0.0025 UJ	0.000538 UJ	0.000538 UJ	0.000538 UJ	0.032 U
	3 (FR)		0.00146 UJ	NA	NA	NA	NA	0.062 U
W003	1	peristaltic	0.0009	0.0025 UJ	0.00049 UJ	0.00049 UJ	0.00049 UJ	0.091 J
	2		0.0018	0.0025 U	0.0005 U	0.0005 U	0.00050 U	0.65
	3		0.0019 J	0.0025 UJ	0.000505 UJ	0.000505 UJ	0.000505 UJ	0.03 U
W004	1	peristaltic	0.0025	0.00842 J	0.0005 U	0.0005 UJ	0.00050 U	0.029 U
	2		0.0043	0.0162 J	0.000485 U	0.000485 U	0.000485 U	0.03 U
	2 (FR)		NA	0.0135	0.000485 U	0.000485 U	0.000485 U	NA
	3		0.00188 J	0.0025 UJ	0.000526 UJ	0.000526 UJ	0.000526 UJ	0.074 U
W006	1	peristaltic	0.0028	0.00256 U	0.0005 U	0.0005 UJ	0.00050 U	0.068 J
	2		0.0018	0.00252 U	0.00051 U	0.00051 U	0.00051 U	0.53 U
	3		0.002 J	0.0025 U	0.000521 U	0.000521 UJ	0.000521 U	0.031 U
W007	1	peristaltic	0.0015	0.0026 U	0.000495 U	0.000495 UJ	0.000495 U	0.082 J
	2		0.0017	0.00261 U	0.0005 U	0.0005 U	0.00050 U	0.031 U
	3		0.0024 J	0.0025 U	0.000524 U	0.000524 UJ	0.000524 U	0.031 U
W008	1	peristaltic	0.0019	0.0025 U	0.000472 U	0.000472 UJ	0.000472 U	0.13 U
	2		0.0042	0.0025 U	0.00049 U	0.00049 U	0.00049 U	0.031 U
	3		0.0047 J	0.0025 U	0.000515 U	0.000515 UJ	0.000515 U	0.032 U
W009	1	peristaltic	0.0017	0.0025 U	0.0005 U	0.0005 U	0.00050 U	0.07 U
	2		0.0016	0.0025 U	0.000485 U	0.000485 U	0.000485 U	0.03 U
	3		0.0018 J	0.0025 U	0.000532 U	0.000532 UJ	0.000532 U	0.03 U

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Table 2-4. Chemical Concentrations for Round 2 COPCs in Surface Water for Amphibians

Sampling Location	Sampling Event[a]	Sampling Method	Zinc (dissolved; mg/L)	Total PCBs (µg/L)	2,4′-DDT (µg/L)	4,4′-DDT (µg/L)	Total DDTs (µg/L)	4-Chloro-3-methylphenol (µg/L)
W013-1	1	peristaltic	0.0009 J	NA	0.000472 U	0.000472 U	0.000472 U	0.029 U
	2		0.0014	NA	NA	NA	NA	0.074 J
	3		0.00163 UJ	NA	NA	NA	NA	0.033 U
W013-1	1	XAD	NA	0.00334 J	0.000001957 J	0.00000934 J	0.0000654 J	NA
	2		NA	0.0120 J	0.00000349	0.00001244	0.0000931 J	NA
	3		NA	0.00798 J	0.00000541 J	0.00001779	0.000241 J	NA
W013-2 (FR)[c]	1	peristaltic	0.0022 U	NA	0.000526 U	0.000526 U	0.000526 U	0.029 U
	2		0.0018	NA	NA	NA	NA	NA
	1	XAD	NA	0.00251 J	0.000001672 J	0.00000831 J	0.0000552 J	NA
	2		NA	0.00206 J	0.000000457 U	0.000001342 J	0.0000496 J	NA
	3		NA	0.00216	0.0000011 J	0.00000279 J	0.0000857 J	NA
W017	1	peristaltic	0.0016	NA	0.00051 U	0.00051 U	0.00051 U	0.044 U
	2		0.0017	0.00256 U	0.00051 U	0.00051 U	0.00051 U	0.031 U
	3		0.0018 J	0.0025 U	0.000481 U	0.000693 J	0.00190 J	0.063 U
W018[d]	1	peristaltic	0.0049	NA	0.000481 U	0.000481 U	0.000481 U	0.029 U
	2		0.0023	NA	NA	NA	NA	0.32 J
	3		0.00267	NA	NA	NA	NA	0.031 U
	1	XAD	NA	0.000439 J	0.00000103 J	0.000004054 J	0.0000610 J	NA
	2		NA	0.00169 J	0.000007887 J	0.00001278	0.000103 J	NA
	3		NA	0.000830 J	0.000001195 J	0.00000511 J	0.0000999 J	NA
W019	1	peristaltic	0.0016	0.00253 U	0.000481 U	0.000481 U	0.000481 U	0.041 U
	2		0.0017	0.00263 U	0.0005 U	0.00115	0.0024	0.031 U
	3		0.00174	0.0025 U	0.000485 U	0.000485 UJ	0.000485 U	0.031 U
W021	1	peristaltic	0.0032	0.0025 U	0.000481 U	0.000481 U	0.000481 U	0.034 U
	2		0.002	0.0026 U	0.000481 U	0.000481 U	0.000481 U	0.032 U
	3		0.00161	0.0025 U	0.00049 U	0.00049 UJ	0.00049 U	0.031 U

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Table 2-4. Chemical Concentrations for Round 2 COPCs in Surface Water for Amphibians

Sampling Location	Sampling Event[a]	Sampling Method	Zinc (dissolved; mg/L)	Total PCBs (µg/L)	2,4′-DDT (µg/L)	4,4′-DDT (µg/L)	Total DDTs (µg/L)	4-Chloro-3-methylphenol (µg/L)
W022	1	peristaltic	0.0419	0.0025 U	0.00049 U	0.00049 U	0.00049 U	0.046 U
	2		0.0037	0.0025 U	0.00049 U	0.00049 U	0.00049 U	0.029 U
	3		0.0027	0.00717 J	0.0005 U	0.0005 UJ	0.00050 U	0.03 U

[a] Sampling Event 1 – November 2004; Sampling Event 2 – March 2005; Sampling Event 3 – July 2005.
[b] PCB Aroclors were not analyzed at W002 during the November 2004 sampling event.
[c] There is no peristaltic data (only XAD data) for W013-2, Event 3.
[d] PCB Aroclors and pesticides were not analyzed at W018 during the March and July 2005 sampling events using the peristaltic method.

COPC – chemical of potential concern
FR – field replicate
J – estimated concentration
N – presumptive evidence of a compound
NA – not analyzed
T – value calculated or selected from multiple results
U – not detected at given concentration

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Table 2-5. Selected Chronic and Acute Eco SLs for Amphibian Round 2 COPCs

COPC	Unit	Chronic Eco SL	Acute Eco SL	Source	Comments
Zinc (dissolved)	mg/L	0.0365	0.0362	AWQC	Eco SLs are hardness-dependent and were adjusted using AWQC equations to correspond with a hardness of 25 mg/L calcium carbonate (estimate for the LWR). Criteria are for dissolved fraction.
4-Chloro-3-methylphenol	µg/L	0.6	30	ODEQ	Acute Eco SL was divided by a uncertainty factor of 50 to calculate the chronic Eco SL.
Total PCBs	µg/L	0.014	2	ODEQ and Tier II	Total PCBs criterion applies to total PCBs, as either sum of all homologs, Aroclors, or congeners.
2,4′-DDT	µg/L	0.001	1.1	AWQC	AWQC for 4,4'-DDT applies to 2,4'-DDT.
4,4′-DDT	µg/L	0.001	1.1	AWQC	
Total DDTs	µg/L	0.001	1.1	AWQC	AWQC for 4,4'-DDT applies to total DDTs.

AWQC – ambient water quality criteria
COPC – chemical of potential concern
Eco SL – Ecological Screening Level
EPA – US Environmental Protection Agency
LWR – Lower Willamette River
ODEQ – Oregon Department of Environmental Quality
PCB – polychlorinated biphenyl

Table 2-6. Summary of Surface Water Locations Where Round 2 COPCs are Greater than Eco SLs

Round 2 COPC	Location Where Surface Water Concentration > Eco SL	Toxicity Data Uncertainty	Notes
Zinc	W022 only	Amphibian-specific data are lower than Eco SLs; Eco SLs may underestimate risks to amphibians.	Surface water concentration was greater than Eco SL during November 2004 sampling event (which may not be ecologically relevant exposure period for amphibians).
4-Chloro-3-methylphenol	W003 only	Eco SL are based on acute value divided by 50 and may overestimate risks to amphibians.	Surface water concentration was greater than Eco SL during March 2005 sampling event (which likely represents breeding period).
Total PCBs	W004 only	Amphibian-specific data are several orders of magnitude higher than Eco SLs; Eco SLs may overestimate risks to amphibians.	Surface water concentration was greater than Eco SL was during March 2005 sampling event (which likely represents breeding period).
2,4′-DDT	W001 only	Amphibian-specific data are several orders of magnitude higher than Eco SLs; Eco SLs may overestimate risks to amphibians.	Surface water concentration was greater than Eco SL based on NJ-qualified data and was during March 2005 sampling event.
4,4′-DDT	W0019 only	Amphibian-specific data are several orders of magnitude higher than Eco SLs; Eco SLs may overestimate risks to amphibians.	Surface water concentration was greater than Eco SL was during March 2005 sampling event (which likely represents breeding period).
Total DDTs	W001, W017, and W019 only	Amphibian-specific data are several orders of magnitude higher than Eco SLs; Eco SLs may overestimate risks to amphibians.	Surface water concentration was greater than Eco SL based on NJ-qualified data at W001; Surface water concentrations were greater than Eco SLs during March 2005 sampling event for W001 and W019and during July 2005 sampling event for W017.

COPC – chemical of potential concern
Eco SL – ecological screening level
PCB – polychlorinated biphenyl
N – presumptive evidence of a compound
J – estimated concentration

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APPENDIX G: ECOLOGICAL RISK ASSESSMENT

Attachment G8

Screening Assessment for Aquatic Plants

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1.0 INITIAL SCREENING PROCESS

This attachment presents the screening assessment that was conducted to identify chemicals of potential concern (COPCs) for aquatic plants in the Lower Willamette River (LWR). In the initial screening process for aquatic plants, two pathways of exposure and effect were evaluated; surface water and transition zone water (TZW) data were compared to ambient water quality criteria (AWQC) or TRVs. Surface water was the only pathway that is considered complete and significant for aquatic plants and for which some relevant toxicity data were available. TZW is considered an incomplete pathway for aquatic plants; however, per the request of EPA, this pathway was evaluated in this screening assessment. Direct contact exposure of aquatic plants to sediment was not evaluated in this screening assessment due to the lack of appropriate toxicity data available to evaluate risks to aquatic plants. The results of the surface water and TZW screens are presented in Section 2.0 and the further screening evaluation of surface water and TZW Round 2 COPCs and screening conclusions for aquatic plants are presented in Section 3.0.

2.0 WATER ASSESSMENT

In the water assessment, the exposure of aquatic plants was evaluated by comparing detected chemical concentrations in surface water and TZW collected at the Study Area to chemical concentrations in water known to be toxic to aquatic organisms. The results of the surface water screen are presented in Section 2.1. The results of the transition zone water screen are presented in Section 2.2.

2.1 SURFACE WATER SCREEN

This section presents the initial screening process for surface water assessment including screening of surface water data and Round 2 COPC identification.

2.1.1 Selection of COIs

Chemicals of interest (COIs) for aquatic plants were defined as any chemical detected in surface water sampled in the quiescent areas of the Study Area (Table 2-1), excluding crustal elements (i.e., aluminum; see Section 2.0 of Appendix G).

2.1.2 Identification of Round 2 COPCs

Round 2 COPCs were identified for aquatic plants by comparing the maximum detected surface water concentration of each COI to the respective chronic Eco SL. If the maximum concentration was greater than the Eco SL for a given COI, then that COI was identified as a Round 2 COPC. This screening approach is consistent with EPA guidance (EPA 2006).

A review of water quality regulatory benchmarks and literature-based thresholds was conducted to develop the Eco SLs for chemicals in water. Eco SLs were developed for all aquatic plant surface water COIs except individual dioxins and furans other than 2,3,7,8-TCDD because no data were available. Eco SLs, presented in Attachment G3, were developed by LWG and revised based on EPA's comments (EPA 2006). LWG adopted EPA's proposed hierarchy for Eco SL selection (LWG 2006) and in accordance with EPA's comments, a UF of 50 was used to calculate a chronic screening value from an acute screening value when no chronic data were available. There is high uncertainty in the use of a UF of 50, and LWG recommends the use of a range of UFs for the Baseline Environmental Risk Assessment (BERA). The Eco SLs are considered to be protective of all aquatic receptors.

Table 2-2 presents the maximum concentrations, the corresponding chronic Eco SL, and the Round 2 COPC determination for each aquatic plant COI. The following six COIs were identified as surface water Round 2 COPCs for aquatic plants:

- Zinc (dissolved)
- 4-Chloro-3-methylphenol
- Total polychlorinated biphenyls (PCBs)
- 2,4′-DDT
- 4,4′-DDT
- Total DDTs

These Round 2 COPCs in surface water were further evaluated in the screening assessment for aquatic plants.

2.1.3 Further screening evaluation

Further screening of Round 2 surface water COPCs was conducted for aquatic plants. First, a general habitat assessment of the surface water sampling locations for aquatic plants was conducted to provide a general understanding of the overall exposure to aquatic plants across the Study Area and across seasonal changes. Second, a comparison of the surface water Eco SLs to plant-specific toxicity data was conducted.

2.1.3.1 Aquatic plant exposure evaluation

The exposure area for aquatic plants was evaluated in amphibian habitat and quiescent, backwater areas throughout the Study Area, where surface water samples were collected. Sampling methods used for the collection of surface water in the identified amphibian habitat areas were either peristaltic or XAD or both. Some uncertainty may be associated with the data analyzed from the peristaltic samples because these samples were collected over a short period of time and may overestimate exposure concentrations for and risks to aquatic plants. Details on surface water sampling methods are described in Section 2.0 of Appendix G. Concentrations of Round 2 COPCs measured at each sampling location for each sampling event are presented and summarized in Attachment G7.

The habitat at the 14 sample locations falls into two general categories: low-sloping beaches and steeper, riprapped or rocky banks. Descriptions of each sample location are presented in Attachment G7. In general, low-sloping beaches generally have shallower water, aquatic vegetation, and/or large woody debris, whereas steeper, riprapped, or rocky banks are generally adjacent to deeper water and have some aquatic vegetation or shrubs. Aquatic plants, including submerged plants, emergent herbaceous and woody plants, shrubs, and trees were observed at most of these locations during the aquatic plant and amphibian/reptile reconnaissance survey (Integral et al. 2004).

2.1.3.2 Evaluation of Aquatic Plant Toxicity Data

This section identifies the studies or sources from which the Eco SLs used to evaluate aquatic plant Round 2 COPCs were derived. Eco SLs were selected based on the proposed hierarchy detailed in Attachment G3. Table 2-3 presents the chronic and acute Eco SLs for the surface water Round 2 COPCs for aquatic plants. For all aquatic plant Round 2 COPCs, except 4-chloro-3-methylphenol (which was based on water quality criteria protective of aquatic life), Eco SLs were based on toxicity studies that examined effects on invertebrate and fish species. The relevancy of these values to aquatic plant species is unknown. Therefore, a literature search for plant-specific toxicity values was conducted for comparison with the Eco SLs used in this screening assessment.

The Calcasieu Estuary baseline ecological risk assessment identified plant-specific screening values (MacDonald ES 2002, Appendix D), the majority of which are based on the lowest chronic values (LCVs) identified for aquatic plants by Suter and Tsao (1996). LCVs are available for most of the aquatic plant Round 2 COPCs, except for 4-chloro-3-methylphenol, 2,4′-DDT, and 4,4′-DDT. Discussions of the plant-specific toxicity values for each Round 2 COPC are presented below.

Zinc (dissolved)—The LCV identified for zinc by Suter and Tsao (1996) was 0.030 mg/L and was based on the concentration at which the growth of green algae (*Selenastrum capricornutum*) was initially inhibited (Bartlett et al. 1974). This value is not significantly lower than the chronic Eco SL; therefore, risks to aquatic plants based on the Eco SL are similar to what would be predicted by the LCV.

Total PCBs—The LCV identified for total PCBs by Suter and Tsao (1996) was 0.144 µg/L and was based on a reduction in carbon fixation in a 24-hour test on green alga (*Scenedesmus quadricaudata*) as reported by Laird (1973). This value is greater than the chronic Eco SL; therefore, risks based on the Eco SL may be overestimated to aquatic plants.

DDTs—The LCV identified for DDT by Suter and Tsao (1996) was 0.3 µg/L and was based on effects on growth and morphology of the alga *Chlorella vulgaris*, as reported by Sodergren (1968). This value is greater than the chronic Eco SL; therefore, risks based on the Eco SL may be overestimated to aquatic plants.

2.2 TRANSITION ZONE WATER SCREEN

This section presents the screening assessment for the TZW pathway of exposure for the aquatic plant communities of the LWR. The identification of TZW Round 2 COPCs for aquatic plants is the same as that for benthic invertebrates (Attachment G2), in which exposure was evaluated by comparing maximum COI concentrations in TZW to Eco SLs. TZW COIs were identified as any analyte detected in TZW samples collected at a depth ≤ 38 cm, excluding crustal elements (i.e., aluminum, beryllium, cobalt, iron, magnesium, manganese and potassium; see Section 2.0 of

Appendix G). The list of TZW Round 2 COPCs is repeated in Table 3-1. These chemicals were retained as TZW Round 2 COPCs for aquatic plants.

TZW Round 2 COPCs were further evaluated in the benthic risk assessment (Section 3.0 of Appendix G) using the TZW framework to arrive at potential iCOCs for benthic invertebrates. The potential iCOCs identified for benthic invertebrates were evaluated further in this TZW screening assessment for aquatic plants (Table 3-2). Further TZW screening of potential iCOCs in TZW included 1) a general exposure assessment of the surface water sampling locations for aquatic plants was summarized to provide a general understanding of the overall exposure to aquatic plants across the Study Area, and 2) a comparison of the Eco SLs for potential iCOCs in TZW to plant-specific toxicity data.

2.2.1 Aquatic Plant Exposure Evaluation

TZW samples were collected at nine properties throughout the Study Area. Details on the sample locations and a summary of the TZW results are presented in the benthic risk assessment in Section 3.0 of Appendix G. Most of the TZW sampling locations were not located in the quiescent areas identified as plant habitat; and of those TZW sampling locations located in quiescent areas, most had potential iCOC concentrations less than Eco SLs.

2.2.2 Evaluation of Aquatic Plant Toxicity Data

Eco SLs for potential iCOCs in TZW were selected based on the proposed hierarchy detailed in Attachment G3. Table 3-3 presents the chronic and acute Eco SLs for potential TZW Round 2 iCOCs. Most of the acute and chronic values for the potential TZW Round 2 iCOCs are based on toxicity studies that examined effects on invertebrate and fish species, and the relevancy of these values to aquatic plant species is unknown. Therefore, a literature search for plant-specific toxicity values was conducted for comparison with the Eco SLs used in this assessment.

Discussions of the plant-specific toxicity values for each Round 2 COPC in TZW identified from this literature search are presented below. No plant-specific toxicity studies were found for 2-methylnaphthalene, benzo(a)anthracene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenzo(a,h)anthracene, fluorine, indeno(1,2,3-cd)pyrene, 2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDE, or 4,4′-DDT.

Acenaphthene—The LCV identified for aquatic plants by Suter and Tsao (1996) was 520 µg/L and was based on a 96-hour EC50 reduction in cell numbers for the alga *Selenastrum capricornutum*, as reported by EPA (1978). This value is greater than the chronic Eco SL; therefore, the Eco SL for acenaphthene may overestimate risks to aquatic plants.

Anthracene—Djomo et al. (2004) reported an EC50 and no-observed-adverse-effect-level (NOAEL) of 1,040 and 3.0 µg/L, respectively, based on growth inhibition of the green alga *Scenedesmus subspicatus* in a 7-day study. The EC50 value is greater than both the acute and chronic Eco SLs, but the NOAEL is less than the acute Eco SL. Therefore, the Eco SL for anthracene may underestimate or overestimate risks to aquatic plants.

Benzo(a)pyrene—Djomo et al. (2004) reported an EC50 and NOAEL of 1.48 and 0.1 µg/L, respectively, based on growth inhibition of the green alga *Scenedesmus subspicatus* in a 7-day study. The EC50 value is greater than both the acute and chronic Eco SL, but the NOAEL is less than the acute Eco SL. Therefore, the Eco SL for benzo(a)pyrene may underestimate or overestimate risks to aquatic plants.

Fluoranthene—The LCV identified for aquatic plants by Suter and Tsao (1996) was 54,400 µg/L. This value is greater than the chronic Eco SL; therefore, the Eco SL for fluoranthene may overestimate risks to aquatic plants.

Naphthalene—The LCV identified for aquatic plants by Suter and Tsao (1996) was 33,000 µg/L and was based on inhibited cell numbers observed in a 48-hour EC50 with the alga *Chlorella vulgaris*, as reported by EPA (1980). Also, Djomo et al. (2004) reported an EC50 and NOAEL of 68,210 and 4,150 µg/L, respectively, based on growth inhibition of the green alga *Scenedesmus subspicatus* in a 7-day study. All of these values are greater than the chronic Eco SL; therefore, the Eco SL for naphthalene may overestimate risks to aquatic plants.

Phenanthrene—Djomo et al. (2004) reported an EC50 and NOAEL of 50,240 and 2,740 µg/L, respectively, based on growth inhibition of the green alga *Scenedesmus subspicatus* in a 7-day study. These values are greater than the chronic Eco SL; therefore, the Eco SL for phenanthrene may overestimate risks to aquatic plants.

Pyrene—Djomo et al. (2004) reported an EC50 and a NOAEL of 18.72 and 1.44 µg/L, respectively, based on growth inhibition of the green alga *Scenedesmus subspicatus* in a 7-day study. The EC50 value is greater than the chronic Eco SL, but the NOAEL is less than the chronic Eco SL. Therefore, the Eco SL for pyrene may underestimate or overestimate risks to aquatic plants.

DDT—The LCV identified for DDT by Suter and Tsao (1996) was 0.3 µg/L and was based on effects on growth and morphology of the alga *Chlorella vulgaris*, as reported by Sodergren (1968). This value is greater than the chronic Eco SL; therefore, risks based on the Eco SL may be overestimated to aquatic plants.

Cyanide—The LCV identified for aquatic plants by Suter and Tsao (1996) was 30 µg/L and was based on growth inhibition observed for the alga *Scenedesmus quadricauda*, as reported by EPA (1985). This value is greater than the chronic Eco SL; therefore, the Eco SL for cyanide may overestimate risks to aquatic plants.

Perchlorate—Dean et al. (2004) reported an 25% inhibition concentration (IC25) of 615,000 µg/L, based on growth inhibition of the green alga *Raphidocelis subcapitata* in a 96-hour study. This plant toxicity threshold is several orders of magnitude greater

than the chronic Eco SL. Therefore, the chronic Eco SL for perchlorate may overestimate risks to aquatic plants.

3.0 SCREENING ASSESSMENT CONCLUSIONS

The following section presents that uncertainties and screening conclusions for aquatic plants.

3.1 UNCERTAINTIES

There are several uncertainties associated with the amphibian surface water screening assessment, including the following:

- **The ecological relevance of TZW exposure for LWR aquatic plant communities**—The exposure of the aquatic plant communities of the LWR to chemicals in TZW is uncertain but likely varies according to the plant species. In addition, locations of TZW samples are often not located within quiescent habitat areas identified for aquatic plants and may be in deeper waters where rooted aquatic plants are not likely to grow. Because localized exceedances of Eco SLs are limited within the quiescent areas, the likelihood of risk to the community level is low.
- **The use of Eco SLs for aquatic plants**—A literature review of toxicity studies yielded a few sources of toxicity thresholds specifically for plant assessment endpoints. The majority of the reviewed studies indicated that the chronic Eco SLs were more conservative than the reported plant-specific toxicity thresholds. All of these studies examined the effects of growth inhibition on algae. No studies that examined the effects on rooted aquatic vegetation were identified. Therefore, while the relevancy of these algal toxicity thresholds may be uncertain for higher plants, these thresholds suggest that most of the chronic Eco SLs are protective of aquatic plants.
- **Surface water sampling methods**—All of the Round 2 COPC concentrations that are greater than the chronic Eco SLs are based on the surface water results from the peristaltic method. Locations W013 and W018, which were both sampled using the XAD method, do not have concentrations greater than Eco SLs. Some uncertainty may be associated with the data analyzed from the peristaltic samples because these samples were collected over a short period of time and may overestimate exposure concentrations for and risks to amphibians.

- **TZW sampling methods**—Sampling methods used for the collection of TZW included the use of peepers and probes inserted into the sediment for passive or active TZW collection. TZW was collected once at each sampling location; and as a result, the data represent a "snapshot" of TZW quality at each station. Therefore, some uncertainty may be associated with these data with regard to the exposure of aquatic plants.
- **The use of NJ-qualified data**—Surface water concentrations of 2,4′-DDT and total DDTs at W001 were greater than Eco SLs, but were each based on a single NJ-qualified result. 2,4′-DDT is the only isomer that was detected in this sample. NJ-qualification indicates "the presence of an analyte that has been 'tentatively identified,' and the associated numerical value represents its approximate concentration" (EPA 1999). The qualification indicates that the analyst believed that the result was due to analytical interference from a chemical other than the target analyte. The NJ-qualified result is biased high and will result in an overestimation of risk.

3.2 AQUATIC PLANT SCREENING CONCLUSIONS

Locations of TZW samples are often not located within quiescent habitat areas identified for aquatic plants and may be in deeper waters where rooted aquatic plants are not likely to grow and be exposed. Furthermore, because localized exceedances of Eco SLs are limited within the quiescent areas, these chemicals in TZW would not pose a risk to aquatic plants at the community level. Surface water concentrations were greater than the chronic Eco SL at only one surface water location within the Study Area for total PCBs, zinc, 2,4′ DDT, 4,4′-DDT, and 4-chloro-3-methylphenol. Thus, these chemicals in surface water would not pose a risk to amphibians at the community level. Furthermore, plant-specific toxicity data for the majority of the Round 2 COPCs identified for surface water and potential iCOCs in TZW (including total PCBs, total DDTs, acenaphthene, fluoranthene, naphthalene, phenanthrene, cyanide, and perchlorate) were greater than the Eco SLs, which suggests that risks based on the Eco SLs may be overestimated for aquatic plants. The aquatic plant community of the LWR consists of species that are expected to exist in the habitat of an industrial harbor providing additional evidence that risks to aquatic plants at the Study Area are not significant at the community level.

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TABLES

Table 2-1. COIs for Aquatic Plants

Chemicals of Interest	
Metals	
Antimony	Nickel
Arsenic	Selenium
Cadmium	Silver
Chromium	Thallium
Copper	Zinc
Lead	
Butyltins	
Butyltin ion	Tributyltin ion
Dibutyltin ion	
PAHs	
2-Methylnaphthalene	Chrysene
Acenaphthene	Dibenzo(a,h)anthracene
Acenaphthylene	Fluoranthene
Anthracene	Fluorene
Benzo(a)anthracene	Indeno(1,2,3-cd)pyrene
Benzo(a)pyrene	Naphthalene
Benzo(b)fluoranthene	Phenanthrene
Benzo(b+j)fluoranthene	Pyrene
Benzo(g,h,i)perylene	
Benzo(k)fluoranthene	
Phthalates	
BEHP	Diethyl phthalate
Butylbenzyl phthalate	Dimethyl phthalate
Dibutyl phthalate	Di-n-octyl phthalate
SVOCs	
1,4-Dichlorobenzene	Hexachlorobenzene
Benzoic acid	Hexachlorobutadiene
Carbazole	Isophorone
Phenols	
4-Chloro-3-methylphenol	Phenol
PCBs	
Aroclor 1242	Aroclor 1260
Aroclor 1248	Total PCBs
Aroclor 1254	
Dioxins and Furans	
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin	1,2,3,6,7,8-Hexachlorodibenzofuran
2,3,7,8-Tetrachlorodibenzofuran	1,2,3,7,8,9-Hexachlorodibenzofuran
1,2,3,7,8-Pentachlorodibenzo-*p*-dioxin	2,3,4,6,7,8-Hexachlorodibenzofuran
2,3,4,7,8-Pentachlorodibenzofuran	1,2,3,4,6,7,8-Heptachlorodibenzo-*p*-dioxin

Table 2-1. COIs for Aquatic Plants

Chemicals of Interest	
1,2,3,7,8-Pentachlorodibenzofuran	1,2,3,4,6,7,8-Heptachlorodibenzofuran
1,2,3,4,7,8-Hexachlorodibenzo-*p*-dioxin	1,2,3,4,7,8,9-Heptachlorodibenzofuran
1,2,3,6,7,8-Hexachlorodibenzo-*p*-dioxin	Octachlorodibenzo-*p*-dioxin
1,2,3,7,8,9-Hexachlorodibenzo-*p*-dioxin	Octachlorodibenzofuran
1,2,3,4,7,8-Hexachlorodibenzofuran	
Herbicides	
Dalapon	
Pesticides	
Aldrin	Endosulfan sulfate
cis-Chlordane	Endrin
trans-Chlordane	Endrin ketone
Total Chlordane	Heptachlor
2,4′-DDD	Heptachlor epoxide
2,4′-DDE	alpha-HCH
2,4′-DDT	beta-HCH
4,4′-DDD	gamma-HCH
4,4′-DDE	delta-HCH
4,4′-DDT	Methoxychlor
Total DDTs	cis-Nonachlor
Dieldrin	trans-Nonachlor
alpha-Endosulfan	Oxychlordane
beta-Endosulfan	

BEHP – bis(2-ethylhexyl) phthalate
COI – chemical of interest
HCH – hexachlorocyclohexane
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound

Table 2-2. Round 2 COPCs Identified in Surface Water for Aquatic Plants

COI	Unit	Maximum Surface Water Concentration	Chronic Eco SL	Round 2 COPC?[a]
Metals[b]				
Antimony (total)	mg/L	0.000125 T	0.03	no
Arsenic (dissolved)	mg/L	0.00064	0.15	no
Cadmium (dissolved)	mg/L	0.00003	0.00009	no
Chromium (dissolved)	mg/L	0.00033	0.0238	no
Copper (dissolved)	mg/L	0.00164 J	0.00274	no
Lead (dissolved)	mg/L	0.000152	0.00054	no
Nickel (dissolved)	mg/L	0.00111	0.0161	no
Selenium (total)	mg/L	0.0009 J	0.005	no
Silver (dissolved)	mg/L	0.000061	0.0001	no
Thallium (total)	mg/L	0.000013 J	0.04	no
Zinc (dissolved)	mg/L	0.0419	0.0365	**yes**
Butyltins				
Butyltin ion	µg/L	0.02 J	0.072	no
Dibutyltin ion	µg/L	0.0073 J	0.072	no
Tributyltin ion	µg/L	0.0023 J	0.072	no
PAHs				
2-Methylnaphthalene	µg/L	0.062	2.1	no
Acenaphthene	µg/L	0.0262	23	no
Acenaphthylene	µg/L	0.012 J	306.9	no
Anthracene	µg/L	0.011 J	0.73	no
Benzo(a)anthracene	µg/L	0.012 J	0.027	no
Benzo(a)pyrene	µg/L	0.0075 J	0.014	no
Benzo(b)fluoranthene	µg/L	0.009 J	0.6774	no
Benzo(b+j)fluoranthene	µg/L	0.0023505 J	0.6415	no
Benzo(g,h,i)perylene	µg/L	0.00185	0.4391	no
Benzo(k)fluoranthene	µg/L	0.008 J	0.6415	no
Chrysene	µg/L	0.0089 J	2.042	no
Dibenzo(a,h)anthracene	µg/L	0.0072 J	0.2825	no
Fluoranthene	µg/L	0.082	6.16	no
Fluorene	µg/L	0.021	3.9	no
Indeno(1,2,3-cd)pyrene	µg/L	0.0057 J	0.275	no
Naphthalene	µg/L	0.0169	12	no
Phenanthrene	µg/L	0.096	6.3	no
Pyrene	µg/L	0.047	10.11	no
Phthalates				
BEHP	µg/L	0.025 J	3	no
Butylbenzyl phthalate	µg/L	0.027 J	3	no
Dibutyl phthalate	µg/L	0.00217 J	3	no

Table 2-2. Round 2 COPCs Identified in Surface Water for Aquatic Plants

COI	Unit	Maximum Surface Water Concentration	Chronic Eco SL	Round 2 COPC?[a]
Diethyl phthalate	µg/L	0.00233	3	no
Dimethyl phthalate	µg/L	0.0000344 J	3	no
Di-n-octyl phthalate	µg/L	0.000142 J	3	no
SVOCs				
1,4-Dichlorobenzene	µg/L	0.019 J	15	no
Benzoic acid	µg/L	2.2 J	42	no
Carbazole	µg/L	0.03 J	18.6	no
Hexachlorobenzene	µg/L	0.00701	3.68	no
Hexachlorobutadiene	µg/L	0.00256	9.3	no
Isophorone	µg/L	0.018 J	130	no
Phenols				
4-Chloro-3-methylphenol	µg/L	0.65	0.6	**yes**
Phenol	µg/L	0.17 J	110	no
PCBs				
Aroclor 1242	µg/L	0.00007044 NJ	0.053	no
Aroclor 1248	µg/L	0.00705 J	0.081	no
Aroclor 1254	µg/L	0.00657	0.033	no
Aroclor 1260	µg/L	0.008232 NJ	94	no
Total PCBs	µg/L	0.0162 J	0.014	**yes**
Dioxins and Furans				
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin[c]	pg/L	0.26	100	no
Herbicides				
Dalapon	µg/L	0.23	2	no
Pesticides				
Aldrin	µg/L	0.000002143 J	0.3	no
cis-Chlordane	µg/L	0.00104 J	0.0043	no
trans-Chlordane	µg/L	0.00108 J	0.0043	no
Total chlordane	µg/L	0.00293 J	0.0043	no
2,4′-DDD	µg/L	0.0000468	0.001	no
2,4′-DDE	µg/L	0.00000317 J	0.001	no
2,4′-DDT	µg/L	0.0187 NJ	0.001	**yes**
4,4′-DDD	µg/L	0.0001072	0.001	no
4,4′-DDE	µg/L	0.0000609	0.001	no
4,4′-DDT	µg/L	0.00115	0.001	**yes**
Total DDTs	µg/L	0.0199 NJ	0.001	**yes**
Dieldrin	µg/L	0.00004427 J	0.056	no
alpha-Endosulfan	µg/L	0.000569	0.056	no
beta-Endosulfan	µg/L	0.0000331	0.056	no
Endosulfan sulfate	µg/L	0.00077 J	0.056	no
Endrin	µg/L	0.000000787 J	0.036	no

Table 2-2. Round 2 COPCs Identified in Surface Water for Aquatic Plants

COI	Unit	Maximum Surface Water Concentration	Chronic Eco SL	Round 2 COPC?[a]
Endrin ketone	µg/L	0.0000011 J	0.036	no
Heptachlor	µg/L	0.000000141 J	0.0038	no
Heptachlor epoxide	µg/L	0.000004911 J	0.0038	no
alpha-HCH	µg/L	0.000076215 J	0.08	no
beta-HCH	µg/L	0.00000832 J	0.08	no
gamma-HCH	µg/L	0.000768	0.08	no
delta-HCH	µg/L	0.00167 NJ	0.08	no
Methoxychlor	µg/L	0.0111	0.03	no
cis-Nonachlor	µg/L	0.000002263 J	0.0043	no
trans-Nonachlor	µg/L	0.00000929 J	0.0043	no
Oxychlordane	µg/L	0.00000267 J	0.0043	no

[a] Round 2 COPCs were identified where the maximum detected surface water concentration was greater than the chronic Eco SL.

[b] Concentrations of metals were evaluated as total or dissolved concentrations, depending on the Eco SL.

[c] 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin was evaluated as a representative for all dioxins and furans.

BEHP – bis(2-ethylhexyl) phthalate
COI – chemical of concern
COPC – chemical of potential concern
Eco SL – ecological screening level
HCH – hexachlorocyclohexane
J – estimated concentration
N – presumptive evidence of a compound
PCB – polychlorinated biphenyl
SVOC –semivolatile organic compound
PAH – polycyclic aromatic compound
T – value calculated or selected from multiple results
Bold identifies where the maximum detected surface water concentration was greater than the chronic Eco SL.

Table 2-3. Selected Chronic and Acute Eco SLs for Aquatic Plant Round 2 COPCs in Surface Water

Round 2 COPC	Unit	Chronic Eco SL	Acute Eco SL	Source	Comments
Zinc (dissolved)	mg/L	0.0365	0.0362	AWQC	Eco SLs are hardness-dependent and were adjusted using AWQC equations to correspond with a hardness of 25 mg/L calcium carbonate (estimate for the LWR). Criteria are for dissolved fraction.
4-Chloro-3-methylphenol	µg/L	0.6	30	ODEQ	Acute Eco SL was divided by an uncertainty factor of 50 to calculate the chronic Eco SL.
Total PCBs	µg/L	0.014	2	ODEQ and Tier II	Total PCBs criterion applies to total PCBs, as either sum of all homologs, Aroclors, or congeners.
2,4′-DDT	µg/L	0.001	1.1	AWQC	
4,4′-DDT	µg/L	0.001	1.1	AWQC	
Total DDTs	µg/L	0.001	1.1	AWQC	AWQC for 4,4′-DDT applies to total DDTs.

AWQC – ambient water quality criteria
COPC – chemical of potential concern
Eco SL – ecological screening level
EPA – US Environmental Protection Agency
LWR – Lower Willamette River
ODEQ – Oregon Department of Environmental Quality
PCB – polychlorinated biphenyl

Table 3-1. Round 2 COPCs Identified in TZW for Aquatic Plants

Round 2 Chemicals of Potential Concern	
Metals	
Barium (total)[a]	Nickel (dissolved)[b]
Cadmium (dissolved)[b]	Sodium (total)[a]
Copper (dissolved)[b]	Vanadium (total)[a]
Lead (dissolved)[b]	Zinc (dissolved)[b]
PAHs	
2-Methylnaphthalene	Chrysene
Acenaphthene	Dibenzo(a,h)anthracene
Anthracene	Fluoranthene
Benzo(a)anthracene	Fluorene
Benzo(a)pyrene	Indeno(1,2,3-cd)pyrene
Benzo(b)fluoranthene	Naphthalene
Benzo(g,h,i)perylene	Phenanthrene
Benzo(k)fluoranthene	Pyrene
SVOCs	
1,2-Dichlorobenzene	Dibenzofuran
1,4-Dichlorobenzene	
Pesticides	
2,4′-DDD	4,4′-DDE
2,4′-DDT	4,4′-DDT
4,4′-DDD	Total DDTs
Herbicides	
Dalapon	Silvex™
VOC	
1,1-Dichloroethene	Ethylbenzene
1,2,4-Trimethylbenzene	Isopropylbenzene
1,3,5-Trimethylbenzene	m,p-Xylene
Benzene	o-Xylene
Carbon disulfide	Total xylenes
Chlorobenzene	Toluene
Chloroethane	Trichloroethene
cis-1,2-Dichloroethene	Vinyl chloride
Cyanide	
Cyanide	
Perchlorate	
Perchlorate	

[a] Eco SL is based on a total concentration; dissolved concentrations were not evaluated.
[b] Eco SL is based on a dissolved concentration; total concentrations were not evaluated.
COPC – chemical of potential concern
PAH – polycyclic aromatic hydrocarbon

SVOC – semivolatile organic compound
TZW – transition zone water
VOC – volatile organic compounds

Table 3-2. Potential TZW iCOCs for Aquatic Plants

Potential iCOC	
PAHs	
2-Methylnaphthalene	Chrysene
Acenaphthene	Dibenzo(a,h)anthracene
Anthracene	Fluoranthene
Benzo(a)anthracene	Fluorene
Benzo(a)pyrene	Indeno(1,2,3-cd)pyrene
Benzo(b)fluoranthene	Naphthalene
Benzo(g,h,i)perylene	Phenanthrene
Benzo(k)fluoranthene	Pyrene
Pesticides	
2,4′-DDD	4,4′-DDE
2,4′-DDT	4,4′-DDT
4,4′-DDD	Total DDTs
Cyanide	
Cyanide	
Perchlorate	
Perchlorate	

iCOC – initial chemicals of concern
PAH – polycyclic aromatic hydrocarbon
TZW – transition zone water

Table 3-3. Selected Acute and Chronic Eco SLs for Potential iCOCs in TZW

Round 2 COPC	Acute Eco SL (µg/L)	Chronic Eco SL (µg/L)	Source	Comments
PAHs				
2-Methyl-naphthalene	37	2.1	Tier II	The Eco SLs for 1-methynapthalene were used as a surrogate.
Acenaphthene	80	23	Tier II	The acute and chronic Eco SLs are the EPA calculated final acute value (FAV) and final chronic value (FCV), respectively, for sediment quality guideline development.
Anthracene	13	0.73	Tier II	Eco SLs were calculated for AWQC, though were never adopted.
Benzo(a)anthracene	0.49	0.027	Tier II	
Benzo(a)pyrene	0.24	0.014	Tier II	
Benzo(b)fluoranthene	NA	0.6774	EPA (2003)	Per EPA (EPA 2006), the acute Eco SL was not adopted.
Benzo(g,h,i)perylene	NA	0.4391	EPA (2003)	Per EPA (EPA 2006), the acute Eco SL was not adopted.
Benzo(k)fluoranthene	NA	0.6415	EPA (2003)	Per EPA (EPA 2006), the acute Eco SL was not adopted.
Chrysene	NA	2.042	EPA (2003)	Per EPA (EPA 2006), the acute Eco SL was not adopted.
Dibenzo(a,h)anthracene	0.24	0.2825	Tier II (acute); EPA (2003) (chronic)	Acute Eco SL for benzo(a)pyrene was used as a surrogate.
Fluoranthene	33.6	6.16	Tier II	Eco SLs were calculated for AWQC, though were never adopted.
Fluorene	70	3.9	Tier II	
Indeno(1,2,3-cd)pyrene	0.24	0.275	Tier II (acute); EPA (2003) (chronic)	Acute Eco SL for benzo(a)pyrene was used as a surrogate.
Naphthalene	190	12	Tier II	
Phenanthrene	30	6.3	Tier II	Eco SLs were calculated for AWQC, though were never adopted.
Pyrene	NA	10.11	EPA (2003)	Per EPA (EPA 2006), the acute Eco SL was not adopted.
Pesticides				
2,4′-DDD	1.1	0.001	AWQC	AWQC for 4,4'-DDT applies to 2,4'-DDD.
2,4′-DDT	1.1	0.001	AWQC	AWQC for 4,4'-DDT applies to 2,4'-DDT.
4,4′-DDD	1.1	0.001	AWQC	AWQC for 4,4'-DDT applies to 4,4'-DDD.
4,4′-DDE	1.1	0.001	AWQC	AWQC for 4,4'-DDT applies to 4,4'-DDE.
4,4′-DDT	1.1	0.001	AWQC	
Total DDTs	1.1	0.001	AWQC	AWQC for 4,4'-DDT applies to total DDTs.

Table 3-3. Selected Acute and Chronic Eco SLs for Potential iCOCs in TZW

Round 2 COPC	Acute Eco SL (µg/L)	Chronic Eco SL (µg/L)	Source	Comments
Cyanide				
Cyanide	22	5.2	AWQC	
Perchlorate				
Perchlorate	NA	18	Goleman et al. (2002)	Chronic Eco SL was based on amphibian growth LOAEL following 70 days of exposure.

AWQC – ambient water quality criteria
COPC – chemical of potential concern
Eco SL – ecological screening level
EPA – US Environmental Protection Agency
FAV – final acute value
FCV – final chronic value
iCOC – initial chemicals of concern
NA – not applicable
PAH – polycyclic aromatic hydrocarbon
SL – screening level
TZW – transition zone water